Appendix H

Part 5

developed in Bursian 2003 (NOAEL and LOAEL). This assessment was previously sent to U.S. EPA (Viacom 2004) and is included here as Appendix B.

- The TRVs used to assess toxicity for kingfishers are all from other bird species. Gallinaceous birds, such as chickens and pheasants are among the most sensitive birds with respect to exposures to PCBs and other dioxin-like chemicals. The assessment fails to evaluate recent scientific evidence that kingfishers may be much less sensitive than any of these birds. This evidence is found in the Housatonic field studies (Henning and Brooks, 2003). At the Housatonic site, where PCB levels in fish are one to two orders of magnitude greater than they are at this site, there was no noticeable impact to breeding success of native kingfishers near the site. While the original PCBs disposed of at the Housatonic site may be different from those disposed of at Neal's Landfill, it has been established that the TEQs are the most accurate predictor of toxicity to fish eating birds. The TEQs of the Housatonic fish when evaluated as mg TEQs/kg PCBs are similar, but higher, than those found at the Neal's Landfill site. This large discrepancy between predicted risk to reproduction and actual field measured reproduction shows how relying on a single line of evidence such as theoretical hazard quotients can lead to improper conclusions.
- There is a growing body of evidence that suggests that it is inappropriate to apply chicken based TRVs to piscivorous birds for TEQs. For example, studies over the last several years by a number of researchers indicate that predatory birds, such as bald eagles, ospreys, and kingfishers are much more resistant to the effects of dioxins and related chemicals (Woodford et. al., 1998; Kennedy et. al. 2003; Henning and Brooks, 2003). Elliott and Harris (2002) noted the following based on their extensive review of data for piscivorous birds:

    - *In summary, the results of this study are consistent with the emerging data from both field and laboratory studies which indicate that predatory birds are not particularly sensitive to some of the effects of TCDD. Assessments based on field studies on eagles (Elliott et al., 1996) and ospreys (Woodford et al., 1998) and the comparative egg injection work with kestrels, indicate that raptors are rather insensitive to some of the toxic and biochemical effects of TCDD and PCBs. Elliot et al., (1996) suggested a no-effect level (based on hepatic CYP1A in hatchlings) of 100 pg/g TEQs and a lowest effect level of 303 pg/g.*

- In summary, the raptor TEQ egg LOAEL ranges conservatively from 210 pg/g to 303 pg/g based on enzyme induction, which is not typically an endpoint chosen for an ERA since it is not an ecologically relevant endpoint and occurs at much lower concentrations than effects on reproduction and development. Furthermore, since these values are based on the induction of enzymes, it is likely that effect levels in predatory birds for ecologically relevant endpoints, such as reproductive and developmental endpoints, are much greater. If these TRVs were used in the U.S. EPA's ERA, then the predicted risks to piscivorous birds, such as kingfishers, would be substantially less, and if other TRVs were available for such raptors based on endpoints such as reproductive/developmental endpoints, the predicted risk would again be much less.

- CBS has previously provided U.S. EPA TRVs for the Great Blue Heron (Viacom 2004) and this analysis is provided again in Appendix A. Since the Great Blue Heron is also a predatory piscivore, the TRVs derived for the Great Blue Heron are likely more representative for the Kingfisher. CBS therefore recommends that the TRVs in Viacom 2004 for the Great Blue Heron be used for the kingfisher rather than those the U.S. EPA has derived based on Gallinaceous birds. If this substitution were made, the calculated hazard quotients would drop to about those shown for the "LOAEL" kingfisher case in U.S. EPA's risk assessment. Those hazard quotients approximate 1 based on the 2003 data set and show little risk to avian piscivores.

Response 53: Eight sets of TRVs are used in the FERA, the comments pertain to 2 of these 8 sets. The TRVs in question are based on meta-analysis of multiple toxicity studies, a procedure used to derive mink dietary no effect (500 µg total PCB/kg diet) and low effect (600 µg/kg) TRVs, and kingbird ingestion dose no effect (400 µg total PCB/kg_{BW}-d) and low effect (500 µg/kg_{BW}-d) TRVs (a second set of kingbird PCB ingestion TRVs used in the FERA are not based on meta-analysis). The remaining 5 sets of TRVs in the FERA are derived through other approaches.

The methods used for the meta-analysis are not novel. The approach is based on Leonards, et al. (1995), who used meta-analysis to interpolate mink tissue-based PCB TRVs on a dioxin-equivalent (TEQ) basis. The method used in the FERA for normalizing data from multiple studies to combine them into a single meta-analysis is the same as used by Leonards, et al. (1995). Other examples of the same normalization approach for meta-analysis of ecotoxicological studies include Isnard, et al. (2001), Tanaka and Nakanishi (2001), and Calabrese (2005). The main differences between the methods in the FERA and Leonard, et al. (1995) are minor ones made for site-specific objectives. The FERA meta-analysis TRVs are derived for PCBs on an individual Aroclor basis, instead of TEQs; exposure to mink is quantified on a dietary basis, instead of tissue accumulation; a different regression method is used (adapted from U.S. EPA guidance on effluent toxicity testing); and, consistent with Superfund practice, TRVs are based on the range between no adverse effects and the onset of adverse effects, while the Leonards, et al. (1995) TRVs are based on a high incidence of adverse effects (affecting 50 % of exposed mink).

Following a suggestion of CBS (then Viacom), an uncertainty analysis has been performed for the two sets of meta-analysis TRVs. The procedure, recommended by Dr. John Giesy, ENTRIX, consultant to CBS, is to remove data points individually from the combined data set to assess the effect of incompatibilities between studies and treatments on the TRV interpolation. The results show that the TRVs for PCB dietary exposure to mink, and PCB ingestion dose to birds, are robust to data variability in the upper range of exposures, but not in the lower range of exposures. In other words, the "actual" TRVs are unlikely to be higher than the values used in the FERA–the analysis recommended by CBS resulted in no more than a 20 % increase in the calculated TRVs, and mostly less than 10 % changes related to variability in the upper range of exposures, but the "actual" TRVs might be lower than derived for the FERA–the procedure resulted in 35 to 90 % decreases in calculated TRVs related to variability in the lower range of exposures. This implies that risk calculations based on these TRVs are unlikely to overestimate risk, but the possibility that risk might be underestimated cannot be ruled out.

Meta-analysis refers to techniques for combining the results of multiple studies into a single analysis. CBS incorrectly states that the TRVs derived through meta-analysis are extrapolated, but the meta-analytical method in the FERA is restricted solely to interpolation within the combined data sets, and extrapolation beyond the bounds of the empirical data is not allowed. Incompatibilities between studies because of differences in study design or other factors are potentially important limitations of meta-analysis and, therefore, were evaluated as part of the meta-analysis performed for the ERA.

As discussed in the ERA, multiple studies with several species of mammals show that the reproductive toxicity of PCBs increases with length of exposure or with number of generations continuously exposed.

CBS incorrectly states that Brunström et al. (2001) evaluated only two doses. Brunström et al. (2001) evaluated three doses (control, A50 low, and A50 high), and reported results separately for PCB exposures over 1 and 2 breeding seasons (6 and 18 months, respectively). For the meta-analysis, the single breeding season results were supplemented with one additional high dose reported by Kihiström, et al. (1992), for a total of 4 doses. There are sufficient data to derive 2 TRVs (no effect TRV and low effect TRV) for each of the two exposure durations.

The mink PCB TRV is based on the number of live kits per mated female, which, as a measure of reproductive capacity, is an ecologically relevant endpoint. Reproductive failure in ranch mink fed Great Lakes fish was an early indication that contamination (later shown to be PCBs) in the Great Lakes posed a health risk to mammals.

Brunström et al. (2001) reported whelping rate and litter size of live kits for both 6- and 18-month exposures, from which live kits per mated female can be calculated. For the 18-month exposure A50 high treatment, whelping rate and live litter size decreased by 58 and 55 %, respectively, compared to the control treatment, both of which are statistically significant differences. However, for the same dose with 6-months exposure, there were no more than 5 % differences in whelping rate and live litter size with the control treatment, which are not statistically discernible differences. The sole difference is exposure duration. The combined effect on whelping rate and live litter size resulted in more than a 4-fold decrease in the number of live kits per mated female after 18-months exposure compared to 6 months exposure in the A50 high treatment, for an overall 80 % reduction in live kits per mated female compared to the control treatment at the end of the experiment.

As CBS points out, kit survival at 2 weeks was not reported for the 6-month exposure groups, so this endpoint cannot be used to compare the relative effects of 6- and 18-

month exposures in the Brunström et al. (2001) study. However, comparison of kit survival in long-term and short-term exposure groups is possible in the Restum, et al. (1998) study.

Kit survival is inversely related to exposure duration in the Restum, et al. (1998) study. A dietary dose of 1.0 ppm PCB resulted in statistically decreased kit survival with 6-months exposure, but a dietary dose of 0.5 ppm PCB resulted in statistically decreased kit survival with 16- to 18-months exposure (at kit 3 and 6 weeks age) or with exposure over 2 consecutive generations (at birth, 3 and 6 weeks).

A site-specific mink feeding study was performed for the Housatonic River Superfund site (Bursian, et al. 2006). The dietary concentration of the treatment resulting in decreased kit survival (3.7 mg PCB/kg diet) is higher than the LOAEC TRVs used at other Superfund sites, but resulted in high kit mortality (54 %). The investigators performed probit regression analysis to calculate the dietary concentrations lethal to 20 % and 10 % of kits (LC_{20} and LC_{10}, respectively) and the associated 95 % confidence intervals (CI). The LC_{20} is 1 mg PCB/kg diet (CI: 0.5 – 1.9 mg/kg), and the LC_{10} is 0.2 mg PCB/kg diet (CI: 0.03 – 0.5 mg/kg) (rounded values based on Bursian, et al. 2006). The Bursian, et al. (2006) LC_{20} differs from the FERA LOAEC by less than a factor of 2, reasonably consistent with the observed difference in toxicity between PCB exposure over 1 breeding season versus exposure over 2 breeding seasons. In contrast, the Bursian, et al. (2006) LC_{10} is lower (more conservative) than the FERA NOAEC. However, the 95 % confidence intervals for the Bursian, et al. (2006) LC_{20} and LC_{10} include the values of the LOAEC (0.6 mg PCB/kg diet) and NOAEC (0.5 mg/kg) TRVs, respectively, used in the FERA.

CBS included the relative potency for only one of the treatments in the Bursian study in their comment, which is anomalously high compared to the other four exposure treatment groups (relative potencies from 9.3 to 10.3 mg TEQ/kg PCB). The average relative potency of all five of the exposure treatment groups in Bursian, et al. (2006) is 11.2 mg TEQ/kg PCB, which is lower (less toxic) than any of the data from Neal's Landfill reported in CBS Table 1. The mean relative potency of the Bursian, et al. (2006) study is less than one-half of the mean relative potency for all species collected from Richland Creek at Vernal Pike in May and November 2003 (CBS Table 1), and is only one-third of the relative potency of all species collected at that location by CBS in November 2005 (32.7 mg TEQ/kg PCB). The mean relative potency of the Bursian, et al. (2006) study is approximately two-thirds of the mean relative potency for all species collected from Conard's Branch in May and November 2003 (CBS Table 1), but is less than one-half of the relative potency of creek chub collected at that location by CBS in November 2005 (25.2 mg TEQ/kg PCB). Contrary to CBS's comment that TRVs based on Bursian, et al. (2006) would be "likely conservative" for Neal's Landfill, comparisons of mean relative potencies provide additional evidence that the mixture of PCBs released to the Housatonic River is much less toxic than the mixture released to Conard's Branch and Richland Creek. This means that the PCB TRVs for the Housatonic River site are not adequately protective for the Neal's Landfill site.

With regards to the comment on probit analysis, apparently CBS does not consider the Bursian, et al. (2006) study to be the "highest quality, most relevant study from which TRVs can be determined" when the analyses do not conform with CBS's objectives. The Bursian, et al. (2006) LOAEC resulted in high kit mortality by 6 weeks, which corresponds to nearly a 50 % decrease in the number of live kits at 6 weeks per mated female compared to the control treatment. This is a severe effect, therefore, the onset of adverse effects is expected to occur at some lower dose. The probit analysis performed by Bursian, et al. (2006) is an approach for calculating the dietary PCB exposure associated with the onset of adverse effects on kit mortality. It is obvious from longer-term studies with mink (and other mammals exposed to PCBs), that the results of toxicological studies involving exposures over a single breeding season will underestimate the reproductive toxicity of PCBs associated with exposures over more than one breeding season or to consecutive generations.

Chicken are known to be sensitive to PCBs, and chicken PCB toxicity data are used to derive one set of TRVs to represent higher sensitivity to PCBs, but a second set of TRVs based on doves is also used in the FERA to represent middle sensitivity to PCBs, which brackets uncertainty over the sensitivity of kingfisher to PCBs. It would be inappropriate to solely assess potential risk to kingfisher based only on TRVs for insensitive species when the sensitivity of kingfisher is not known.

The mink ERA submitted by CBS demonstrates that risk to mink may be avoided by use of TRVs based on the single-breeding season exposure study performed by Bursian, et al. (2006). As discussed above, the assertion that the relative potency of the feed used in the Bursian, et al. (2006) mink study is equivalent to that in fish near Neal's Landfill is based on incomplete and selective use of the Bursian, et al. (2006) data (only the relative potency of the one treatment with anomalously high potency is reported by CBS). Use of the mean relative potency of all of the Bursian, et al. (2006) treatments shows much lower relative potency compared to fish near Neal's Landfill, with as much as a 2- to 3-fold difference. Bursian, et al. (2006) also reported LC_{10} and LC_{20} TRVs, which bracket the probable onset of adverse effects. Use of these TRVs result in LC_{10} HQs of 8-9 for mean exposure, and 10-12 for UCL exposure; and LC_{20} HQs of 2 for mean exposure, and 3 for UCL exposure.

Data from a dioxin study with pheasant is used for dose-based TEQ TRVs. Although pheasant are also a gallinaceous species, pheasant is less sensitive to dioxin than chicken. One of CBS's consultants described pheasant as "one of the more tolerant species" to dioxin-like effects (Giesy, et al. 1995; see also Bowerman, et al. 1995).

The kingfisher field study performed at the Housatonic River site is limited by several shortcomings in design, including an insufficiently broad exposure gradient, lack of a control or reference population, and a method of evaluation that is subject to confounding because the results of the field study are compared to that of a single study from the literature for a different location. According to the Housatonic River ERA (USACE/U.S. EPA 2004):

> "The belted kingfisher study results do not definitively support the conclusions of low risk because the data collected in the study are limited. There are several conclusions drawn by the authors that are not strongly supported by the information presented in the report. The conclusion that the kingfisher population is consistent with the quality of habitat present is speculative. ... It introduces significant uncertainties to conclude that the Housatonic River kingfishers fall within the range reported for other kingfisher populations when only one study is referenced."
>
> "... EPA was not provided with an opportunity to review standard operating procedures or protocols prior to receiving the study report ..."
>
> "There were, however, several shortcomings that resulted in uncertainties regarding the conclusions of the study. No information was provided regarding nest search intensity, important endpoints like clutch size and hatching success were not measured, and there were too few visits to the nests during the reproductive cycle. These shortcomings limit the ability to draw rigorous conclusions from the field study results."
>
> "The lack of reference area and small sample size (i.e., n=6) with the statistics used is considered a major source of uncertainty in the kingfisher field study."
>
> "The approach used to estimate dose in the belted kingfisher study had a number of shortcomings. ... The lack of reference area concentrations further compounds uncertainties in the dose gradient. Thus, there are major uncertainties in the dose gradient achieved by this approach, as it is likely too narrow to detect a significant dose-response relationship and the dose associated with a given nest is unknown."
>
> "The kingfisher field study had limited ability to detect differences in reproductive success."

For these reasons, the Housatonic kingfisher field study is not considered an adequate study for reducing uncertainty over the relative sensitivity of kingfisher to PCBs.

CBS's comment on the differences between TEQ toxicity in chicken and "piscivorous birds" is based primarily on effects on raptors (eagles, osprey, kestrels), the sole exception, the aforementioned Housatonic kingfisher field study, is addressed above. The comments regarding raptors are not germane because kingfisher is not a closely related to raptors. Assuming kingfisher sensitivity to PCBs is similar to that of raptors is as uncertain as assuming it is similar to that of chicken. The ERA assesses risk to kingfisher based on a range of sensitivities to address this uncertainty. Also, in a review of avian studies of dioxin-like toxicity performed by U.S. EPA, chicken was not shown to be unusually sensitive:

> "A conclusion of these analyses is that the domestic chicken is, as is generally recognized, the most sensitive tested species, but it is not aberrantly sensitive.

Given the wide range of sensitivities within birds and within mammals to dioxin-like chemicals, test data for chickens should be used." (U.S. EPA 2003).

The review also compared TRVs derived through the species sensitivity distribution (SSD) approach for laboratory versus field studies. The egg TEQ TRVs are lower (showing greater toxicity) based on field studies compared to TRVs based on laboratory studies, even when chicken are included in the laboratory SSD (U.S. EPA 2003). This indicates that the results of chicken studies are not necessarily overprotective for wild birds, and may even be insufficiently protective in some situations.

As discussed in the ERA, the particular values of the egg TEQ TRVs used in the ERA are based on enzyme induction, but the TRVs were chosen because they represent a middle range between the values reported in multiple field studies that resulted in reproductive impacts *as measured in the field*, which are ecologically relevant endpoints.

Kingfisher and heron are not closely related, so there is no basis for concluding that "the TRVs derived for the Great Blue Heron are likely more representative for the Kingfisher". CBS recommends that the great blue heron TRVs be used for kingfisher "rather than those the U.S. EPA has derived based on Gallinaceous birds", but CBS is really asking that the range of TRVs based both on chick and dove studies be replaced with the proposed great blue heron TRVs. The relative sensitivity of kingfisher to PCBs or dioxin-like effects compared to great blue heron is not known, therefore U.S. EPA considers the use of the range of TRVs in the ERA appropriate because it reflects uncertainty over kingfisher sensitivity.

The ERA for great blue heron submitted by CBS shows potential risk to herons feeding in Conard's Branch and the upper portion of Richland Creek based on TEQ in heron eggs. Although the submitted ERA shows very high hazard quotients for dietary TEQ exposure based on the NOAEL TRV, no assessment was made of potential risk on a dietary TEQ LOAEL basis. This leaves this aspect of the heron ERA in the same situation as for kingfisher, risk to dietary TEQ exposure is difficult to assess because species-specific LOAEL data are lacking for great blue heron. The results of the submitted great blue heron are contradictory, risk appears to be acceptable on a PCB basis, but unacceptable on a TEQ basis.

Comment 54: The World Health Organization (WHO) has recently re-issued its recommended mammalian TEFs for PCBs (WHO 2005). The new TEFs should be used for the assessment of risk in the Neal's Landfill risk assessment.

Response 54: Use of the revised World Health Organization mammalian TEFs has no effect on the risk characterization for mink because of approximately parallel changes in both exposure estimates and TRVs. Mean exposure and risk to mink in 2001, 2002, and 2003 were recalculated with the WHO_{2005} mammalian TEFs presented in van den Berg, et al. (2006). Mink dietary exposure on a TEQ basis in Conard's Branch and the upper reach of Richland Creek decreased by 30 % across the 3 years (range of -24 to -36 % for the combined Conard's Branch/Richland Creek exposure scenario). The overall effect on risk also depends on recalculation of the TEQ TRVs with the revised TEFs. The mink TEQ TRVs used in the ERA are based on the geometric mean of the TRVs in two long-term mink studies (Restum, et al. 1998; Brunström, et al 2001). The congener data for the Restum, et al. (1998) study are reported in Tillitt, et al. (1996). The WHO_{2005} TEQs of the three exposure treatments in Restum, et al. (1998) are 24 % lower than the WHO_{1998} TEQs, and the WHO_{2005} TEQs of the two exposure treatments in Brunström, et al. (2001) are 53 % lower than the WHO_{1998} TEQs. The revised mink dietary WHO_{2005} TEQ NOAEC is 2.8 pg/g (compared to 4.6 pg/g WHO_{1998} TEQ), and the LOAEC is 11 pg/g (compared to 18 pg/g WHO_{1998} TEQ), for an overall -39 % decrease in TEQ TRVs. The net effect is a small increase in several mink NOAEC HQs and few mink LOAEC HQs, but, because of rounding of the final HQs, the majority of the mink LOAEC HQs and the rest of the mink NOAEC HQs are unchanged.

CBS has not complied with their own comment. The TEQs in the mink ERA submitted by CBS are calculated with WHO_{1998} TEFs.

Comment 55: Conard's Branch is a very shallow narrow stream. Since kingfishers are dive feeding birds and the water is so shallow, it is unlikely that kingfishers would use this stream reach extensively. U.S. EPA has acknowledged verbally that the vast majority of Conard's Branch is too shallow during non-storm conditions for the dive feeding Kingfisher. But U.S. EPA has stated that adequate water depth would be available during storm periods and that the PAC analysis shows that even short periods of feeding during storms can result in estimated unacceptable risk for the Kingfisher. However, U.S. EPA has failed to take into account that Kingfishers need clear water to locate their prey. During storm periods, the water depth will increase, but the turbidity of the water also increases dramatically (this is very obvious by reviewing the TSS data from any of the numerous storm events CBS has monitored in the stream) decreasing the ability of the kingfisher to see fish in the stream. CBS continues to recommend that a more appropriate avian receptor for Conard's Branch is the Great Blue Heron. Additionally, there is some direct toxicity data for Great Blue Heron and PCBs (see Viacom 2004). Therefore use of this receptor at this location would reduce the uncertainty in the avian risk estimate. CBS has shown a representative risk assessment for great blue herons at the NLF site and thus has developed appropriate factors (feeding range, ingestion rates, body weight, TRVs etc in Viacom 2004 and in Appendix A).

Response 55: Belted kingfishers depend on shallow water for catching prey. Reportedly, the majority of prey are caught within 5 to 6 inches below the surface (Prose 1985). Historically, belted kingfishers have nested along Conard's Branch. A kingfisher burrow was observed in 1997 in the bank of Conard's Branch downstream of the Vernal Pike overpass and upstream of the confluence with Richland Creek (Dan Sparks, USFWS, pers. comm.).

Comment 56: U.S. EPA has calculated theoretical hazard quotients greater than 1 for both mink and kingfisher for the upper reaches of the study area. The hazard quotients are driven by the amount of time for each receptor that they are assumed to forage from Conard's Branch. Conard's Branch is a small short stream that can only support foraging part time for at most 1 mink and/or kingfisher. U.S. EPA is supposed to evaluate ecological risk at the community or population level (U.S. EPA 1999). While the U.S. EPA risk assessment does develop scenarios that show theoretical hazard quotients greater than 1 for these receptors based on very conservative assumptions, U.S. EPA does not show how such a small area could have any substantive impact on the population of either receptor in the broader context of the Richland Creek area.

Response 56: The OSWER directive refers to local populations at or near the site, not regional populations. According to U.S. EPA (1999):

> "The goal of the Superfund program is to select a response action that will result in the recovery and/or maintenance of healthy *local* populations/communities of ecological receptors that *are or should be present at or near the site. ...* Contaminated media that are expected to constrain the ability of *local* populations and/or communities of plants and animals to recover and *maintain themselves in a healthy state at or near the site* (e.g., contamination that significantly reduces diversity, increases mortality, or diminishes reproductive capacity) should be remediated to acceptable levels." [emphases added].

One of the reasons for assessing risk to mink and kingfisher is because protection of piscivorous wildlife from PCB-related risks is expected to be protective of fish and other aquatic organisms. The same rationale is used in setting the PCB federal ambient water quality criteria for environmental effects. Therefore, populations of fish may be at risk of adverse effects in the stream reaches in which mink and kingfisher are potentially at risk. The potential risk to fish in Conard's Branch and the upper portion of Richland Creek as implied in the risk findings for mink and kingfisher are supported by the additional assessment of direct risk to fish in the ERA and by the adverse effects on creek chub growth and survival in the field study by Henshel, et al. (2006).

Comment 57: The U.S. EPA has ignored the latest two data sets for PCB in fish (collected in November 2003 and November 2005) in its estimates of exposure. It should be noted that the average PCB content in fish at station B in the fall of 2005 was 2.27 ppm and at station D .4 ppm. These 2005 averages are below the stated goals for fish at these locations in U.S. EPA's proposed plan.

Response 57: The ERA for the Neal's Landfill has served its purposes for characterizing risk and calculating goals for reducing risk to ecological receptors. It will not be endlessly revised with each additional round of sampling. Data collected subsequent to completion of the ERA should be used for compliance monitoring and trend analysis.

There are pronounced seasonal fluctuations in PCB concentrations in fish related to large fluctuations in fish lipid contents. Fish lipid contents are very low in November, and PCB concentrations are correspondingly at their minima. Since fish lipid contents are much higher in the spring and summer, PCB concentrations are also greater during spring and summer. For this reason, the November fish PCB data sets under-represent exposures of fish, mink, or kingfisher to PCBs, and compliance of November whole-body fish PCB averages with stated goals is an incomplete evaluation of whether risk objectives have been attained.

See Attachment 4 for the November 2005 fish sample data. The November 2005 average PCB content in creek chub at Conard's Branch (station B) is approximately one-half of the concentration at this station in November 2002 (4.05 to 4.89 ppm depending on two approaches for Aroclor analysis). However, the mean TEQ in Conard's Branch creek chub in November 2005 (53 pg/g) is only 25 % less than the November 2002 value (71 pg/g). The inconsistency between PCB and TEQ trends may be related to the change in PCB analytical methods between the sampling periods – Aroclor analysis in 2002 versus the sum of congeners method in 2005.

Evaluation of the average PCB concentration in station D (Richland Creek at Vernal Pike) fish in November 2005 is complicated by the absence of white sucker. In contrast, the average PCB concentration in fish at this station in November 2002 included data for white sucker. Trends should therefore be assessed for the two species collected in both sample events. This shows essentially no trend for PCBs, but a small increase in TEQ in 2005 compared to 2002. The mean PCB content in creek chub in 2005 (0.25 ppm) is barely below the 2002 range (0.27 to 0.33 ppm), but the mean TEQ content in 2005 (8 pg/g) is slightly higher than in 2002 (7 pg/g). The mean PCB content in longear sunfish in 2005 (0.55 ppm) is within the 2002 range (0.48 to 0.59 ppm), but the mean TEQ content in 2005 (16 pg/g) is higher than in 2002 (14 pg/g).

Risk estimates for the November 2005 data at the Richland Creek at Vernal Pike location is misleading without white sucker data. Risk may be approximated by assuming the same empirical relationship between mean concentrations in longear sunfish and white sucker as observed in November 2002. This is the same approach used to estimate missing crayfish data. In November 2002, mean PCB concentration in white sucker was 2.5 times that in longear sunfish, and mean TEQ was 1.8 times. With this approach, November 2005 mink risk estimates for PCBs are in the same range as November 2002: no effect to low effect HQs of 1 to 0.8, and NOAEL to LOAEL HQs for TEQ are slightly higher: 3 to 0.8. The risk estimates for Conard's Branch are lower than in November 2002, but still indicate risk: PCB HQ of 3 (no effect and low effect are the same because of rounding), and NOAEL to LOAEL HQs for TEQ of 8 to 2. The combined foraging scenario over Conard's Branch and Richland Creek for November 2005 shows somewhat smaller risk than in November 2002 for PCBs: no effect to low effect HQs of 2 to 1, but similar risk for TEQ: no effect to low effect HQs of 4 to 1. These are risk estimates for mean exposures, risk estimates for UCL exposures have not been recalculated.

Comment 58: U.S. EPA states in section 6.1 that there is no obvious trend to the fish data. CBS disagrees. U.S. EPA should reconsider this statement in light of the complete fish data set (through fall 2005). As U.S. EPA notes, evaluations of trends in fish data can be confounded by a number of factors. CBS has evaluated trends in both the spring water PCB levels (which ultimately drive PCB trends in fish), and in the PCB data in fish. These analyses were sent to U.S. EPA in CBS 2006a and CBS 2007. These analyses show that indeed there is a downward trend to both the concentration of PCBs in the spring water and the fish. These analyses have been reviewed by U.S. EPA consultants and to the best of our knowledge U.S. EPA has concurred with CBS's conclusions on trend.

Response 58: U.S. EPA concurs with the trend analysis for spring water PCB levels, but considers the trend for fish to be more uncertain. The greater uncertainty in fish PCB trends arises partly because of the much smaller data base for fish samples compared to water samples, and partly because of the greater complexity of the biological and ecological processes affecting bioaccumulation of PCBs in fish compared to the physical processes affecting PCB levels in water. The statement that there is no obvious trend in the fish data included in the ERA is a valid observation, which is reinforced by the contradictory results of comparing mean TEQ in November fish samples between 2002 and 2005 – Conard's Branch data indicate a declining trend, but Richland Creek data indicate no or even increasing trends.

Comment 59: The U.S. EPA has added an assessment of risk to fish. The measure of effects is taken from a Giesy study on dioxin (TCDD) effects on rainbow trout. CBS has several comments on this approach:

Comment 59a: First, U.S. EPA has used a study on rainbow trout. There are no rainbow trout in the NLF drainage and the characteristics of the stream are such that trout will not inhabit this stream (it is a warm water fishery). Rainbow trout have shown to be very sensitive to the effects of dioxin and dioxin like chemicals.

Response 59a: See Response 52.

Comment 59b: Second, U.S. EPA is using a TRV that is based on exposure to dioxin. This ignores the large body of literature available on PCBs. This increases the uncertainty in the analysis and is unnecessary since studies directly assessing the effect of PCBs on fish are available.

Response 59b: U.S. EPA is not aware of controlled long-term PCB exposure studies to adult fish.

Comment 59c: The U.S. EPA conclusion that fish may be at risk in the NLF drainage appears to be at odds with the available fish population data that shows a diverse and plentiful population of fish both in the NLF drainage and at other Bloomington area sites with comparable or higher levels of PCBs (such as Stout's Creek).

Response 59c: As discussed in Response 52, the field study by Henshel, et al. (2006) shows adverse effects on creek chub survival and growth in Conard's Branch and the upper portion of Richland Creek, and in Clear Creek.

Comment 60: The U.S. EPA has added an assessment of a road killed mink reportedly found near the NLF site to the risk assessment. CBS has several comments on this approach:

Comment 60a: First, there is no way to verify where and when the mink was actually found. The mink was reportedly found by a local resident who was also a local environmental activist. This resident reportedly found the mink and then kept it in their freezer for some length of time before handing it over to USFWS for analysis.

Response 60a: The uncertain origin of the road-killed mink is addressed in the ERA. The agreement between modeled and measured accumulation of PCBs or TEQ in mink liver for that location is supporting evidence that the mink foraged in that location. The implication that the mink may have been planted would also require a sophisticated ability to either collect a mink from some other exposure area with PCB levels equivalent to, but not notably higher or lower than the PCB levels along that portion of Richland Creek, or an even more highly sophisticated ability to spike the mink carcass with a quite precise amount of PCBs to give the appearance of local accumulation. While neither scenario can be absolutely ruled out, they are also highly improbable.

Comment 60b: Second, the U.S. EPA analysis of the mink data assumes lipid values for certain key literature to evaluate effects. The use of assumed lipid values is key to their analysis and the validity of these assumed values cannot be verified and is inappropriate.

Response 60b: The use of assumed lipid values in the ERA to allow comparisons of the road-killed mink liver data with published effect levels on both a wet-weight and lipid-normalized basis is fully transparent – assumed values are marked in parentheses, and calculated vales not directly given in the original publications are marked with an asterisk. Comparisons can therefore be easily made with or without derived data.

Comment 60c: Many of the studies evaluated by U.S. EPA have issues which should disqualify them for consideration. For example, many of the studies U.S. EPA cites have endpoints of questionable population level relevance such as enzyme induction, hepatic vitamin A, or mandibular and squamas cell proliferation. The Saginaw Bay studies have been shown to be influenced by other contaminants. The study by Leonard et al (1995) is not a primary source of data and should not be used.

Response 60c: The purpose of the table was to show where the road-killed mink sample falls within a continuum of toxic responses, including no effect concentrations at which no adverse effects are expected. Comparisons were made on the basis of both PCBs and TEQs. TEQs incorporate the dioxin-like effects of co-contaminants into a single value. Leonards, et al. (1995) is included because they derived thresholds for severe adverse impacts on litter size and kit survival based on meta-analysis of multiple toxicological studies of mink, which provides an additional line of evidence for evaluating whether the accumulations of PCBs and TEQ by the road-killed mink were well above LOAECs (the evaluation shows they were not).

Comment 60d. CBS feels the quality assurance issues associated with using a road killed mink supplied by a local activist overwhelm the utility of the data. However, if the data were to be used, the Bursian 2003 study presents data on PCB levels in mink liver that could be used for comparison. The levels of PCB found in the road killed mink liver (77 pg/g) are below levels found in Bursian 2003 livers that may cause population level effects such as kit survival beyond 6 weeks (the threshold from Bursian 2003 for kit survival is given as the geomean of the NOAEL and LOAEL and amounts to 111 pg/g TEQ in the liver of the adult females).

Response 60d: Bursian, et al. (2003) is included in the comparison in the ERA.

Comment 61: CBS has re-calculated the risk for a theoretical heron receptor (Appendix A) and calculated risks for a mink receptor (Appendix B) based on revised TRVs, exposure scenarios and site-use factors that are more reasonable. These calculations indicate that the risk to these two receptors based on average PCB levels in 2003 are minimal, with HQ values only exceeding 1.0 based on the NOAEL, but not for the LOAEL. The greatest HQ based on the NOAEL for mink (based on the 95% UCL concentration in diet) was 1.7 (for PCBs) and 2.7 (for TEQs), which indicates that there is little if any risk to mink. The greatest HQ based on the NOAEL for great blue herons (based on the 95% UCL concentration in diet or modeled concentration in eggs) was 1.0 (for PCBs), which indicates that there is little if any risk to great blue herons. When risks to great blue heron from exposure to TEQ are assessed, there is limited data for piscivores for ecologically relevant endpoints (something more relevant than enzyme induction). From the limited field data, Entrix derived a TRV and calculated a maximum NOAEC-based HQ of 40. This HQ has an uncertain amount of conservatism in it and as a single line of evidence would not support additional remedial actions. As noted above, since PCB levels in fish appear to be declining, it is anticipated that the current risk to receptors is less than even these calculated levels.

Response 61: The ERA for great blue heron submitted by CBS shows potential risk to herons feeding in Conard's Branch and the upper portion of Richland Creek based on TEQ in heron eggs. Although the submitted ERA shows very high hazard quotients for dietary TEQ exposure based on the NOAEL TRV, no assessment was made of potential risk on a dietary TEQ LOAEL basis. This leaves this aspect of the heron ERA in the same situation as for kingfisher, risk to dietary TEQ exposure is difficult to assess because species-specific LOAEL data are lacking for great blue heron. The results of the submitted great blue heron are contradictory, risk appears to be acceptable on a PCB basis, but unacceptable on a TEQ basis.

The mink ERA submitted by CBS demonstrates that risk to mink may be avoided by use of TRVs based on the single-breeding season exposure study performed by Bursian, et al. (2006). As discussed above, the assertion that the relative potency of the feed used in the Bursian, et al. (2006) mink study is equivalent to that in fish near Neal's Landfill is based on incomplete and selective use of the Bursian, et al. (2006) data (only the relative potency of the one treatment with anomalously high potency is reported by CBS). Use of the mean relative potency of all of the Bursian, et al. (2006) treatments shows much lower relative potency compared to fish near Neal's Landfill, with as much as a 2- to 3-fold difference. Bursian, et al. (2006) also reported LC_{10} and LC_{20} TRVs, which bracket the probable onset of adverse effects. Use of these TRVs in the CBS ERA result in LC_{10}

HQs of 8-9 for mean exposure, and 10-12 for UCL exposure; and LC_{20} HQs of 2 for mean exposure, and 3 for UCL exposure.

Comment 62: In summary, the U.S. EPA risk assessment is overly conservative for the following reasons:

a. The U.S. EPA has only calculated theoretical risk based on modeled conservative receptors. This approach is only appropriate to rebut the presumption of risk.
b. U.S. EPA has not performed any field studies to determine if their conservative risk estimates are realistic or to determine if there are any real effects to any populations of receptors.
c. U.S. EPA has not shown how their theoretical risk estimates over the small areas with hazard quotients greater than 1 could manifest into population level effects for any receptors.

U.S. EPA has failed to properly consider the latest studies with mink and piscivorous birds that show less sensitivity with PCBs.

Response 62: All of these comments have been addressed in previous responses.

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ATTACHMENT 1

Technical Memorandum – Model Simulation of Volatilization

QEA
Quantitative Environmental Analysis, LLC
$\partial c/\partial t$

TECHNICAL MEMORANDUM

TO:	Russ Cepko - CBS	**DATE:**	April 2, 2007
FROM:	Kevin Russell	**RE:**	Model Simulation of Volatilization
CC:	Dottie Alke – CBS Jim Rhea – QEA David Glaser – QEA	**JOB#:**	VIAnea:130

A mathematical model of PCB fate and transport was developed and calibrated for two streams (Conard's Branch and Richland Creek) that are impacted by PCBs from the Neal's Landfill Site in Bloomington, IN (QEA 2007). QEA has developed this memorandum to provide additional detail on the simulation of PCB volatilization within the model. This memo describes the mathematical formulation for volatilization in the model and the parameter values that were used to represent this site, and summarizes the results for model-predicted volatilization flux.

Theory and Mathematical Formulation

Volatilization is the process by which PCBs are transported across the air-water interface. Transfer of PCBs at the air-water interface is calculated by the PCB fate sub-model as a function of two parameters: 1) the chemical's Henry's Law Constant; and 2) a mass transfer coefficient that describes the rate by which PCBs are transported across the air-water interface. In the PCB fate sub-model, PCB flux from the water column due to volatilization is expressed based on these parameters as follows:

$$J_v = k_L\left(c_d - \frac{c_{air}}{H^*}\right) \tag{1}$$

where: J_v = volatilization flux ($M\,L^{-2}\,T^{-1}$);
k_L = volatilization mass transfer coefficient ($L\,T^{-1}$);
c_d = freely dissolved PCB concentration in water ($M\,L^{-3}$), which is calculated internally in the model based on the PCB partitioning coefficient – see Section 3.2.1.3.6 in QEA 2007;

www.qeallc.com

305 West Grand Avenue
Suite 300
Montvale, NJ 07645
(201) 930-9890
(201) 930-9805 *fax*

290 Elwood Davis Road
Suite 230
Liverpool, NY 13088
(315) 453-9009
(315) 453-9010 *fax*

80 Glen Street
Suite 2
Glens Falls, NY 12801
(518) 792-3709
(518) 792-3719 *fax*

800 Brazos Street
Suite 308
Austin, TX 78701
(512) 707-0090
(512) 275-0915
fax

c_{air} = vapor phase PCB concentration in air (M L^{-3}); and
H^* = dimensionless Henry's Law Constant (--).

The Henry's Law Constant (*HLC*) equals a chemical's vapor pressure divided by its solubility in water and is calculated from the equilibrium ratio of gas phase and water phase concentrations measured in laboratory experiments. This ratio is a fundamental property of a chemical. A high *HLC* is indicative of a volatile chemical that preferentially accumulates in the air phase, while a low *HLC* is indicative of a non-volatile chemical that preferentially accumulates in the water phase. *HLC* values for PCBs are relatively low compared to other classes of organic chemicals (MacKay et al. 1992). Values of *HLC* are typically presented in units of partial pressure per unit aqueous concentration (e.g., Pa-m^3/mol), while the model uses a dimensionless ratio that is derived from *HLC* by dividing by the product of the universal gas constant and absolute temperature (thus converting pressure into concentration using the ideal gas law):

$$H^* = \frac{HLC}{R(T+273)} \tag{2}$$

where: R = Universal Gas Constant (= 8.314 J/K mol); and
T = water temperature, which is specified in the model inputs (°C).

The volatilization mass transfer coefficient (k_L in Equation 1) in the model is calculated based on the classic "two-film theory" of gas transfer (Whitman 1923; Lewis and Whitman 1924). In this model, k_L is dependent on the rates of mass transfer through relatively thin layers of water and air at the interface, which are in turn dependent on the concentration gradients in the layers, and the diffusivity of PCBs in the layers (O'Connor 1983, 1984):

$$k_L = \frac{k_g k_l}{k_g + \dfrac{k_l}{H^*}} \tag{3}$$

where: k_g = vapor phase mass transfer coefficient (L T^{-1}); and
k_l = water phase mass transfer coefficient (L T^{-1}).

The liquid phase mass transfer coefficient (k_l in Equation 3) is calculated internally by the model as a function of the PCB diffusivity in water and the water depth and current velocity (which are calculated by the hydrodynamic sub-model) using the O'Connor and Dobbins (1958) formulation:

$$k_l = \sqrt{\frac{D_w U}{h}} \tag{4}$$

where: D_w = molecular diffusivity of PCBs in water ($L^2\ T^{-1}$);
h = mean water depth from hydrodynamic sub-model (L); and
U = average current velocity from hydrodynamic sub-model ($L\ T^{-1}$).

Molecular diffusivity for this expression is calculated internally by the model using the equation presented by Hayduk and Laudie (1974):

$$D_w = \frac{13.26x10^{-5}}{\mu^{1.14}(\bar{V})^{0.589}} \tag{5}$$

where: μ = water viscosity (centipoise), which is calculated internally by the model as a function of water temperature; and
$\bar{V}$ = PCB molar volume (cm^3/mol).

Finally, the overall volatilization mass transfer coefficient calculated using Equations 3 through 5 is corrected for temperature effects using the standard Arrhenius Equation (e.g., Chapra 1997):

$$k_L(T) = k_L(20°C)\theta_v^{T-20} \tag{6}$$

where: θ_v = volatilization temperature correction factor (--).

Equation 4 predicts that mass transfer is positively related to current velocity, which reflects the fact that increased turbulence tends to increase the effective surface area of the air-water interface, and thereby the efficiency of gas-liquid exchange. Likewise, Equation 4 predicts that volatilization is inversely related to water depth (i.e., shallower areas will produce more volatilization than deeper areas). Finally, the dependence of k_L on temperature in Equation 6 results in the mass transfer coefficient being positively related to temperature (i.e., volatilization increases at higher temperature).

Application to the Neal's Landfill Site

Based on the equations presented above, the model computes volatilization flux for each computational grid element, dynamically over the course of a simulation. The following site-specific parameters are provided as input to the model: *HLC*, c_{air}, k_g, $\bar{V}$, and θ_v. Values used for these parameters in the Neal's Landfill model were as follows:

- The *HLC* of PCBs varies with chlorination level; a site-specific value was estimated based on published *HLC* values for PCB congeners (Brunner et al. 1990) and was adjusted within the range of values corresponding to Aroclors 1242 and 1248 during model calibration. A value for *HLC* of 5 Pa-m^3/mol, which corresponds to a dimensionless value (H^* in Equation 2) of 0.0021 at 20°C, was used in the model.

- The air phase PCB concentration (c_{air} in Equation 1) was set equal to zero because atmospheric PCB concentrations are typically several orders of magnitude lower than those in water, and thus this value has very little impact on the calculated volatilization flux (e.g., QEA 1999).
- The vapor phase mass transfer coefficient (k_g in Equation 3) was assigned a constant value of 100 m/d, which is a reasonable approximation for streams and rivers because of the limited impact of air motion (winds) on transfer in these systems (O'Connor 1983).
- The molar volume ($\overline{V}$ in Equation 5) was assigned a mean value of 260 cm^3/mol, which is consistent with published molar volumes for the major PCB homologs that compose Aroclors 1242 and 1248 (Mackay et al. 1992).
- The volatilization temperature correction factor (θ_v in Equation 6) was set to a typical value of 1.025 (e.g., Chapra 1997).

The remaining parameters used in the volatilization equations are computed internally by the model as described above (H^*, k_l, D_w, U, h, and μ), or are provided as model inputs (T).

Model Results

The model predictions suggest that volatilization is a relatively minor PCB mass transport process in Conard's Branch and Richland Creek. For example, the model mass balance over the 2001-2005 calibration period indicates that the total volatilization from Conard's Branch was calculated to be 0.09 kg, which represents 3% of the total PCB mass entering from the springs and Spring Treatment Facility effluent (see Figure 3-36 in QEA 2007).

A temporal plot of the model-predicted PCB volatilization rate (i.e., grams per day) is shown with the system flow rate in Figure 1, for years 2001 through 2005. The temporal plots demonstrate the model's calculation of increases in volatilization due to higher current velocities during high flow periods, as well as the generally higher volatilization rates during non-storm conditions in the summertime, due to lower water depths and higher temperatures.

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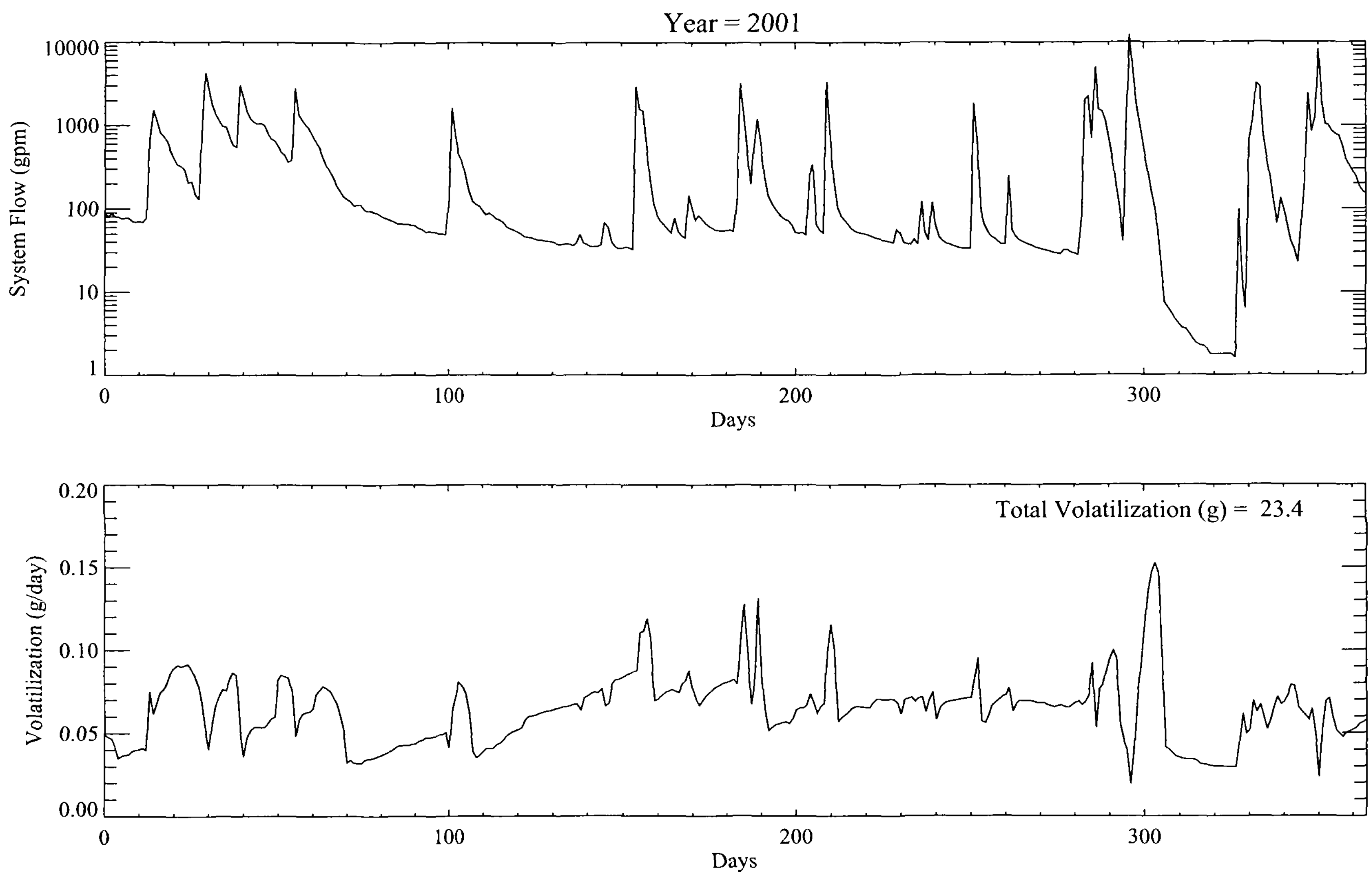

Figure 1a. Temporal profiles of flow and volatilization in Conard's Branch in 2001.

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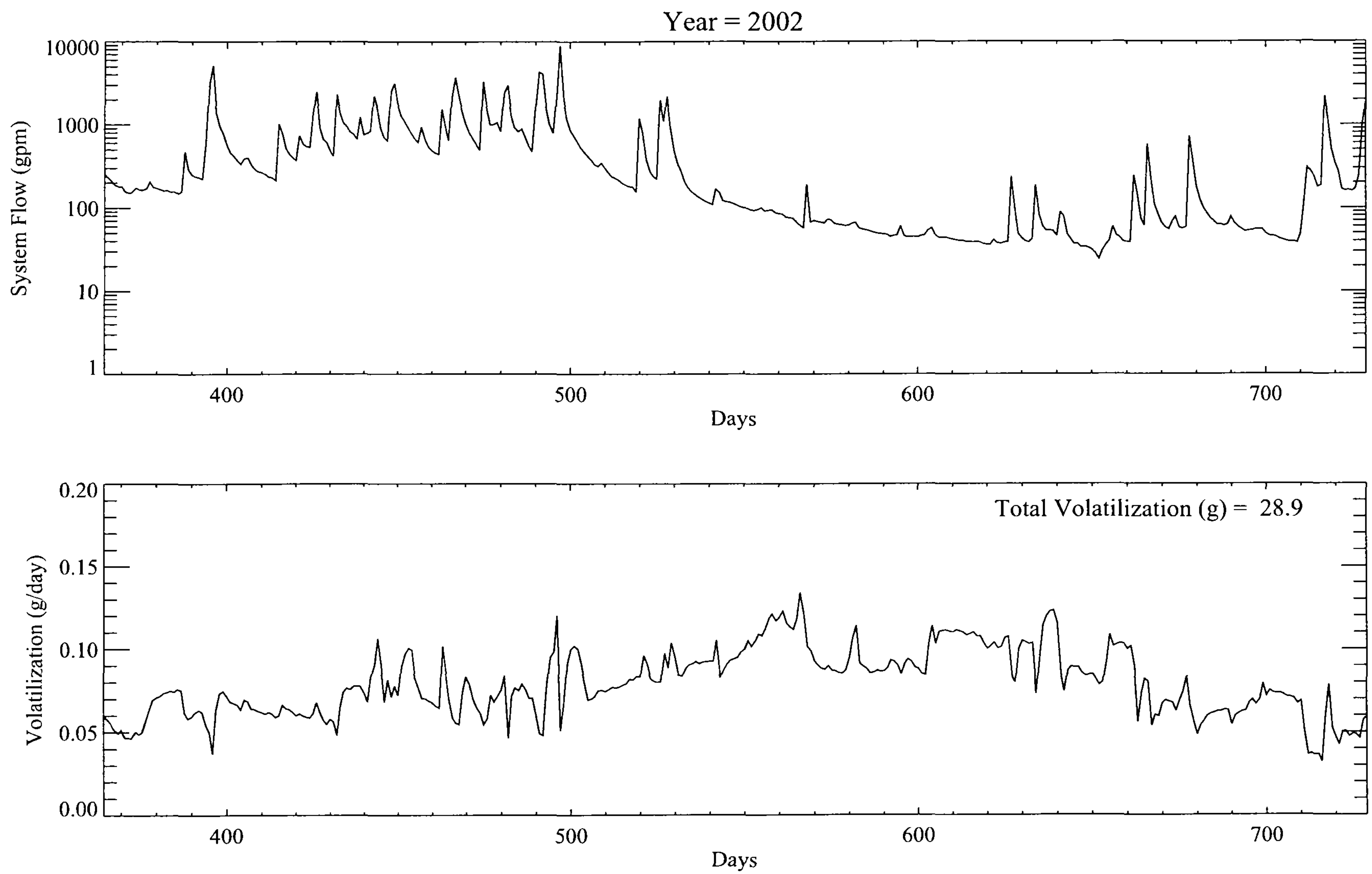

Figure 1b. Temporal profiles of flow and volatilization in Conard's Branch in 2002.

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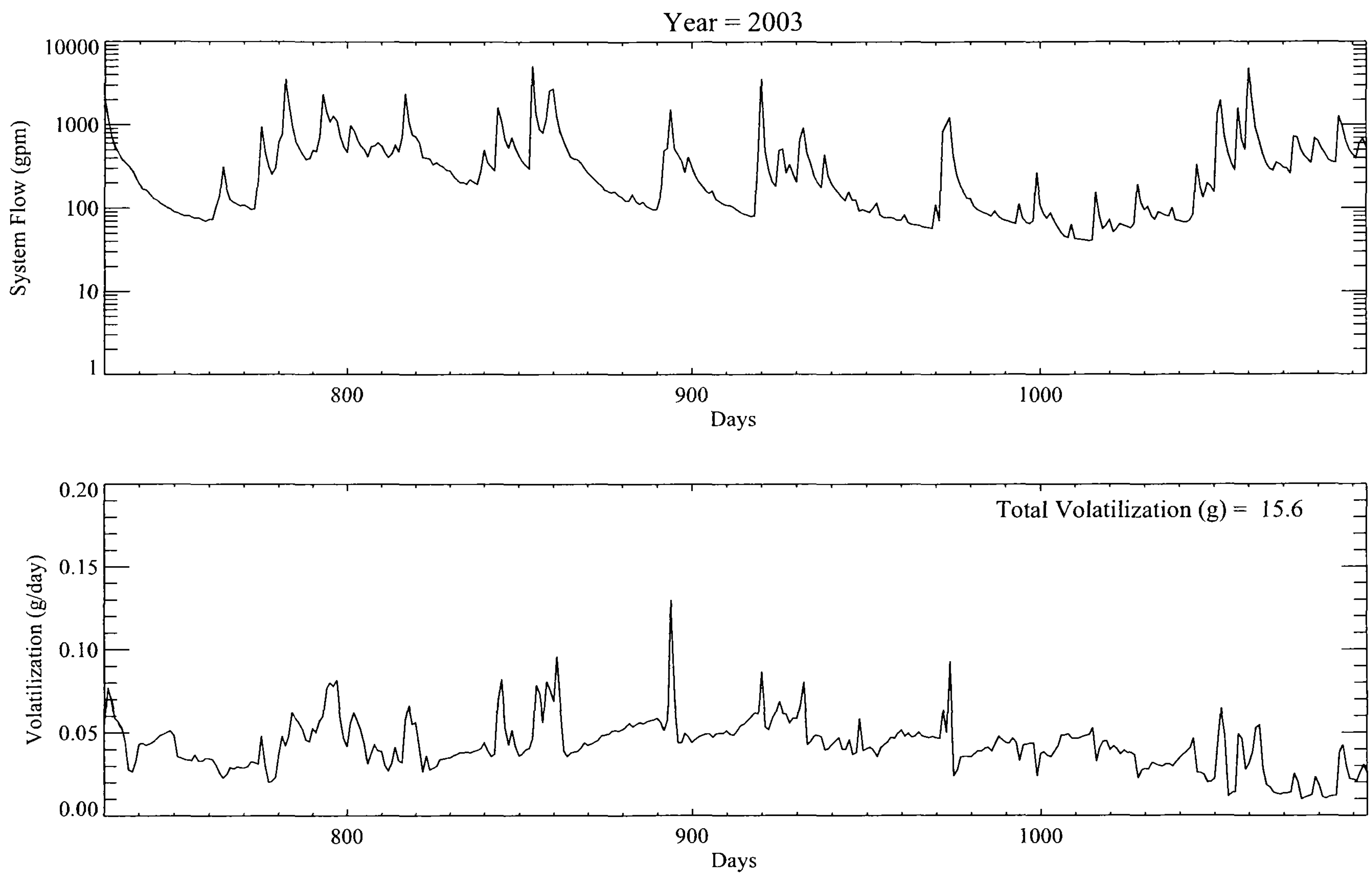

Figure 1c. Temporal profiles of flow and volatilization in Conard's Branch in 2003.

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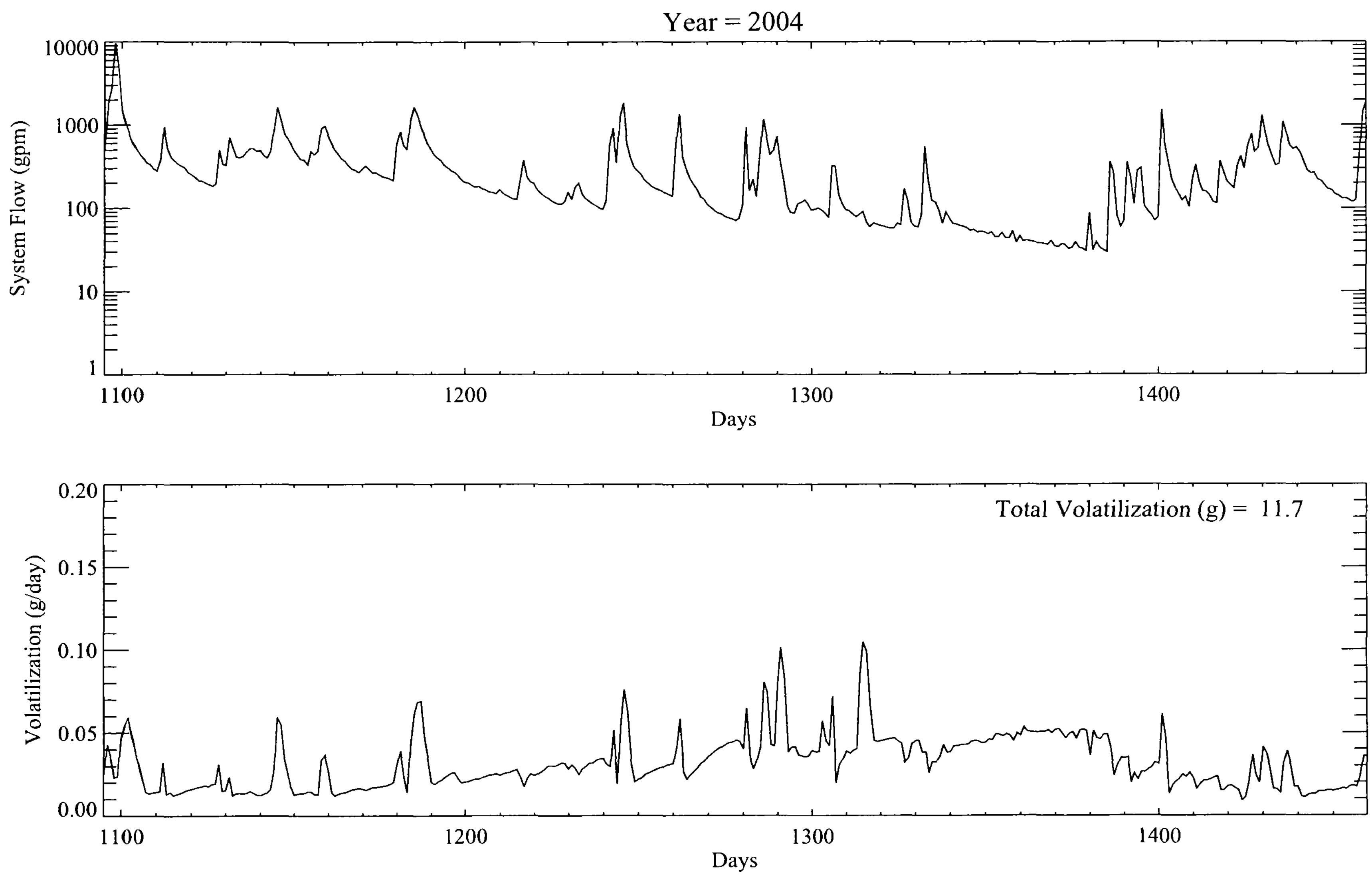

Figure 1d. Temporal profiles of flow and volatilization in Conard's Branch in 2004.

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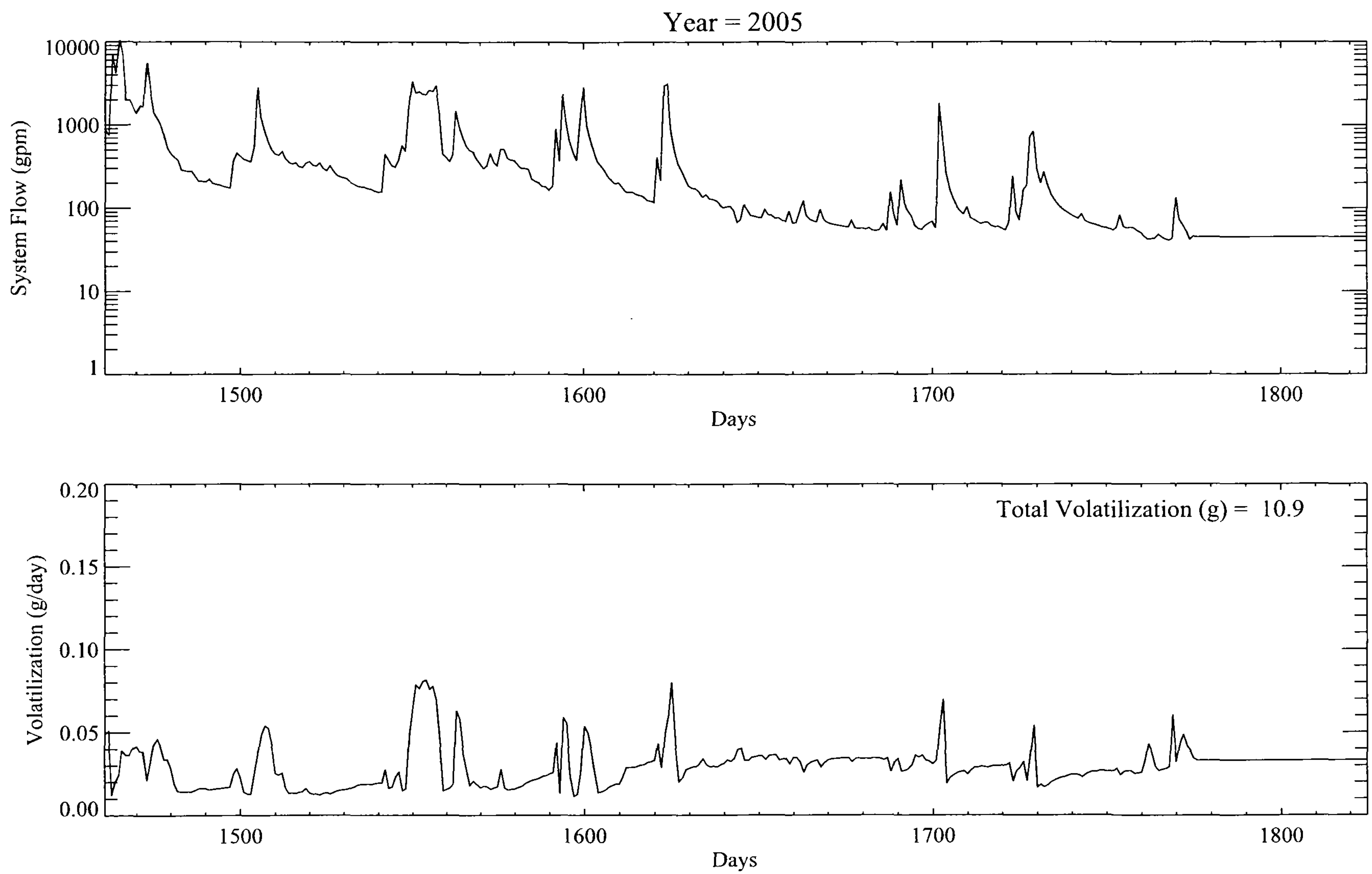

Figure 1e. Temporal profiles of flow and volatilization in Conard's Branch in 2005.

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ATTACHMENT 2

Table 1 – Revised Risk Summary
Table 2 – Revised Risk Summary
EPA's Revised Exposures, Risks, and Hazards
Human Health Risk Assessment
Tables A-1 through Table A-5
Neal's Landfill

TABLE 1

REVISED RISK SUMMARY[a]
HUMAN HEALTH RISK ASSESSMENT
NEAL'S LANDFILL
BLOOMINGTON, MONROE COUNTY, INDIANA

Exposure Pathway		Location[b]									
		Conard's Branch[c]				Richland Creek[d]					
		South Spring	North Spring	Section Pre-A	Section A	Section B	Section C	Section D	Section E	Section F	Section G
Fish Ingestion[e]	Pelagic	6.7E-06	6.7E-06	6.7E-06	6.7E-06	6.7E-06	6.7E-06	5.1E-06	5.1E-06	--	1.0E-05
	Benthic	7.4E-06	7.4E-06	7.4E-06	7.4E-06	7.4E-06	7.4E-06	2.0E-06	2.0E-06	2.7E-06	4.6E-06
Surface Water	Incidental Ingestion	6.3E-09	3.4E-09	4.7E-09	9.6E-10	1.8E-09	1.8E-09	1.1E-09	1.1E-09	--	--
	Dermal Contact	2.9E-06	1.6E-06	2.7E-06	6.6E-07	2.0E-06	2.0E-06	1.5E-06	1.5E-06	--	--
Sediment	Incidental Ingestion	6.7E-08	6.7E-08	6.7E-08	6.7E-08	7.3E-09	7.3E-09	4.3E-09	2.7E-09	3.6E-08	2.1E-08
	Dermal Contact	6.0E-08	6.0E-08	6.0E-08	6.0E-08	6.5E-09	6.5E-09	3.9E-09	2.4E-09	3.2E-08	1.9E-08
Bank Soil	Incidental Ingestion	8.4E-07	8.4E-07	8.4E-07	1.2E-07	3.6E-09	3.6E-09	2.4E-08	2.4E-08	--	--
	Dermal Contact	4.7E-07	4.7E-07	4.7E-07	6.7E-08	2.0E-09	2.0E-09	1.3E-08	1.3E-08	--	--
Floodplain Soil	Incidental Ingestion	3.0E-07	3.0E-07	3.0E-07	1.8E-07	5.6E-08	5.6E-08	--	--	--	--
	Dermal Contact	1.7E-07	1.7E-07	1.7E-07	1.0E-07	3.1E-08	3.1E-08	--	--	--	--
TOTALS - All[f]		2E-05 (80)	2E-05 (86)	2E-05 (81)	2E-05 (94)	2E-05 (87)	2E-05 (87)	9E-06 (82)	9E-06 (82)	3E-06 (98)	5E-06 (99)
TOTALS - No Fish		4E-06	2E-06	3E-06	9E-07	2E-06	2E-06	2E-06	2E-06	7E-08	4E-08

Notes

-- = Not applicable

[a] See Attachment A for detailed receptor- and exposure pathway-specific calculations. All risk values presented are for adult receptors.

[b] See Figures 1, 2 and 3 for locations of Neal' Landfill (including South and North Springs), Conard's Branch (including Sections Pre-A and A), and Richland Creek (including Sections B through G).

[c] *For exposure points in and along Conard's Branch (South and North Springs and Sections Pre-A and A), the totals were calculated as follows.*

South Spring: Fish ingestion risks from the 1-mile site (Section C), surface water risks from South Spring, bank soil and floodplain soil risks from Section Pre-A, and sediment risks for Section A. Note: sediment, bank soil, and floodplain soil risks are each divided by a factor of 3 prior to summing to eliminate overestimation introduced by allowing 100 milligrams per day of each medium when considering each medium on its own.

North Spring: Fish ingestion risks from the 1-mile site (Section C), surface water risks from North Spring; bank soil and floodplain soil risks from Section Pre-A; and sediment risks for Section A. See note for South Spring.

Section Pre-A: Fish ingestion risks from the 1-mile site (Section C), surface water, bank soil, and floodplain soil risks from Section Pre-A; and sediment risks from Section A. See note for South Spring.

Section A: Fish ingestion risks from the 1-mile site (Section C), surface water, sediment, bank soil, and floodplain soil risks from Section A. See note for South Spring.

[d] For exposure points in Richland Creek (Sections B through G), the totals were calculated as follows.

Section B: Fish ingestion and surface water risks from the 1-mile site (Section C); sediment and floodplain soil risks from Section B; and bank soil risks from Section C. See note for South Spring.

Section C: Fish ingestion and surface water risks from the 1-mile site (Section C); sediment and floodplain soil risks from Section B; and bank soil risks from Section C. See note for South Spring.

Section D: Fish ingestion and surface water risks from the 3-mile site (Section E); sediment risks from Section D, bank soil risks from Section E; floodplain soil risks not evaluated for Section D. Note: sediment and bank soil risks are each divided by a factor of 2 prior to summing to eliminate overestimation introduced by allowing 100 milligrams per day of each medium when considering each medium on its own.

Section E: Fish ingestion and surface water risks from the 3-mile site (Section E); sediment and bank soil risks from Section E; floodplain soil risks not evaluated for Section E. See note for Section

Section F: Fish ingestion risks for the 5.5-mile site and sediment risks for Section F, bank soil, surface water, and floodplain soil risks not evaluated for Section F

Section G: Fish ingestion risks for the 12.7-mile site, sediment risk from Section G; surface water, bank soil, and floodplain soil risks not evaluated for Section G

[e] Risks presented represent polychlorinated biphenyl (PCB) risks calculated based on total PCBs.

[f] Value in parentheses represents the percent of total risk represented by risks associated with fish ingestion

TABLE 2

REVISED HAZARD SUMMARY[a]
HUMAN HEALTH RISK ASSESSMENT
NEAL'S LANDFILL
BLOOMINGTON, MONROE COUNTY, INDIANA

Exposure Pathway		Location[b]									
		Conard's Branch[c]				Richland Creek[d]					
		South Spring	North Spring	Section Pre-A	Section A	Section B	Section C	Section D	Section E	Section F	Section G
Fish Ingestion[e]	Pelagic	3.9E-01	3.9E-01	3.9E-01	3.9E-01	3.9E-01	3.9E-01	3.0E-01	3.0E-01	--	6.0E-01
	Benthic	4.3E-01	4.3E-01	4.3E-01	4.3E-01	4.3E-01	4.3E-01	1.2E-01	1.2E-01	1.6E-01	2.7E-01
Surface Water	Incidental Ingestion	3.7E-04	2.0E-04	2.7E-04	5.6E-05	1.0E-04	1.0E-04	6.6E-05	6.6E-05	--	--
	Dermal Contact	1.7E-01	9.2E-02	1.6E-01	3.8E-02	1.2E-01	1.2E-01	8.9E-02	8.9E-02	--	--
Sediment	Incidental Ingestion	3.9E-03	3.9E-03	3.9E-03	3.9E-03	4.3E-04	4.3E-04	2.5E-04	1.6E-04	2.1E-03	1.2E-02
	Dermal Contact	3.5E-03	3.5E-03	3.5E-03	3.5E-03	3.8E-04	3.8E-04	2.3E-04	1.4E-04	1.9E-03	2.0E-03
Bank Soil	Incidental Ingestion	4.9E-02	4.9E-02	4.9E-02	7.0E-03	2.1E-04	2.1E-04	1.4E-03	1.4E-03	--	--
	Dermal Contact	2.7E-02	2.7E-02	2.7E-02	3.9E-03	1.2E-04	1.2E-04	7.9E-04	7.9E-04	--	--
Floodplain Soil	Incidental Ingestion	1.8E-02	1.8E-02	1.8E-02	1.1E-02	3.3E-03	3.3E-03	--	--	--	--
	Dermal Contact	9.8E-03	9.8E-03	9.8E-03	5.9E-03	1.8E-03	1.8E-03	--	--	--	--
TOTALS - All[f]		1.0E+00 (80)	9.5E-01 (86)	1.0E+00 (81)	8.7E-01 (94)	9.4E-01 (87)	9.4E-01 (87)	5.1E-01 (82)	5.1E-01 (82)	1.6E-01 (98)	8.9E-01 (98)
TOTALS - No Fish		2.1E-01	1.3E-01	1.9E-01	5.0E-02	1.2E-01	1.2E-01	9.1E-02	9.1E-02	4.0E-03	1.3E-02

Notes

-- = Not calculated

a See Attachment A for detailed receptor- and exposure pathway-specific calculations. All hazard values presented are for adult receptors

b See Figures 1, 2 and 3 for locations of Neal' Landfill (including South and North Springs), Conard's Branch (including Sections Pre-A and A), and Richland Creek (including Sections B through G)

c For exposure points in and along Conard's Branch (South and North Springs and Sections Pre-A and A), the totals were calculated as follows

South Spring: Fish ingestion hazards from the 1-mile site (Section C), surface water hazards from South Spring, bank soil and floodplain soil hazards from Section Pre-A, and sediment hazards for Section A. Note: sediment, bank soil, and floodplain soil hazards are each divided by a factor of 3 prior to summing to eliminate overestimation introduced by allowing 100 milligrams per day of each medium when considering each medium on its own

North Spring: Fish ingestion hazards from the 1-mile site (Section C), surface water hazards from North Spring, bank soil and floodplain soil hazards from Section Pre-A, and sediment hazards for Section A. See note for South Spring

Section Pre-A: Fish ingestion hazards from the 1-mile site (Section C), surface water, bank soil, and floodplain soil hazards from Section Pre-A, and sediment hazards from Section A. See note for South Spring

Section A: Fish ingestion hazards from the 1-mile site (Section C), and surface water, bank soil, and floodplain soil, and sediment hazards from Section A

d For exposure points in and along Richland Creek (Sections B through G), the totals were calculated as follows

Section B: Fish ingestion and surface water hazards from the 1-mile site (Section C), sediment and floodplain soil hazards from Section B, and bank soil hazards from Section C. See note for South Spring

Section C: Fish ingestion and surface water hazards from the 1-mile site (Section C), sediment and floodplain soil hazards from Section B, and bank soil hazards from Section C. See note for South Spring

Section D: Fish ingestion and surface water hazards from the 3-mile site (Section E), sediment hazards from Section D, bank soil hazards from Section E, floodplain soil hazards not evaluated for Section D. Note: sediment and bank soil hazards are each divided by a factor of 2 prior to summing to eliminate overestimation introduced by allowing 100 milligrams per day of each medium when considering each medium on its own

Section E: Fish ingestion and surface water hazards from the 3-mile site (Section E), and sediment and bank soil hazards from Section E, floodplain soil hazards not evaluated for Section E. See note for Section D

Section F: Fish ingestion hazards for the 5.5-mile site and sediment hazards for Section F, bank soil, surface water, and floodplain soil hazards not evaluated for Section F

Section G: Fish ingestion hazards for the 12.7-mile site, surface water, sediment, bank soil, and floodplain soil hazards not evaluated for Section G

e *Hazards presented represent polychlorinated biphenyl (PCB) hazards calculated based on total PCBs*

f Value in parentheses represents the percent of total hazard represented by hazards associated with fish ingestion

TABLE A-1

REVISED GENERAL AND CHEMICAL-SPECIFIC EXPOSURE PARAMETER VALUES
HUMAN HEALTH RISK ASSESSMENT
NEAL'S LANDFILL
BLOOMINGTON, MONROE COUNTY, INDIANA

General Exposure Parameters					
			Values		
Exposure Parameter	**Acronym**	**Units**	**Adult (18+)**	**Youth (7 to 18)**	**Child (1 to 6)**
Exposure Point Concentrations	EPC	varies	Location Specific		
Exposure Frequency (sw, sed, bks, fs) (NS, SS, Pre-A, A)	EF_{bks}, and EF_{fs}	days/year	20	20	20
Exposure Frequency (sw, sed, bks, fs) (Sections B, C, and D)	EF_{sw}		68	68	68
	EF_{sed}		68	68	68
	EF_{bks}		76	76	76
	EF_{fs}	days/year	76	76	76
Exposure Frequency (sw, sed, bks, fs) (Sections E, F, and G)	EF_{sw}, EF_{sed}, EF_{bks}, and EF_{fs}	days/year	30	30	30
Exposure Frequency (aquatic life)	EF_{al}	days/year	365	365	365
Exposure Duration	ED	years	30	12	6
Body Weight	BW	kg	70	47	15
Ingestion Rate (aquatic life)	IR_{al}	kg/day	see below	NA	NA
Ingestion Rate (sediment and soil)	IR_{sed} and IR_{soil}	mg/day	100	100	200
Ingestion Rate (surface water)	IR_{sw}	L/day	7.50E-03	1.50E-02	1.50E-02
Cooking Reaction Factor	CRF	unitless	0.5	0.5	0.5
Skin Surface Area (sediment)	SA_{sed}	cm^2	2129	1640	809
Skin Surface Area (soil)	SA_{soil}	cm^2	5700	4373	2378
Skin Surface Area (surface water)	SA_{sw}	cm^2	Location Specific		
Adherence Factor (sediment)	AF_{sed}	mg/cm^2	0.3	0.3	0.3
Adherence Factor (soil)	AF_{soil}	mg/cm^2	0.07	0.2	0.2
Dermal Absorption	ABS	unitless	Chemical Specific		
Absorbed Dose per Event	DA_{event}	mg/cm^2-event	Chemical Specific		
Exposure Time	ET	hr/day	1	1	1
Event Frequency	EV	event/day	1	1	1
Fraction Contaminated - Soil	FC	unitless	0.5	0.5	0.5
Conversion Factor 1	CF1	mg/ng	1.00E-06	1.00E-06	1.00E-06
Conversion Factor 2	CF2	mg/µg	1.00E-03	1.00E-03	1.00E-03
Conversion Factor 3	CF3	kg/mg	1.00E-06	1.00E-06	1.00E-06
Averaging Time (carcinogens)	AT_c	days	25550	25550	25550
Averaging Time (noncarcinogens)	AT_{nc}	days	10950	4380	2190
Gastrointestinal Absorption	GI	unitless	Chemical Specific		
Chronic Oral Exposure Reference Dose	RfD	mg/kg-day	Chemical Specific		
Oral Slope Factor	SF	$(mg/kg\text{-}day)^{-1}$	Chemical Specific		

TABLE A-1

REVISED GENERAL AND CHEMICAL-SPECIFIC EXPOSURE PARAMETER VALUES
HUMAN HEALTH RISK ASSESSMENT
NEAL'S LANDFILL
BLOOMINGTON, MONROE COUNTY, INDIANA

Chemical Specific Values[a]									
	Values								
Chemical	ABS	FA	K_p	C_w	τ_{event}	t_{event}	GI	RfD[b]	SF[b]
PCB	0.14	0.5	4.30E-01	see note	11.29	2	0.8 to .96	2.00E-05	2
TEQ	--	--	--	--	--	--	0.5 to 0.83	1.00E-09	1.50E+05
$TEQ_{EPA\ alternate}$	--	--	--	--	--	--	--	--	1.00E+06

C_w = This concentration is the EPC_{sw} in units of mg/cm^3.

a PCB-specific exposure factors were obtained from EPA's RAGS Part E (EPA 2004a).

b PCB-specific toxicity factors were identified as discussed in Sections 4.2.2 and 4.3.2 of the draft HHRA (Tetra Tech 2006).

Ingestion Rate of Aquatic Life					
			Values		
Rate	Fish Type	units	Adult (18+)	Youth (7 to 18)	Child (1 to 6)
1 - Mile	Pelagic	kg/day	0.0060	--	--
	Benthic	kg/day	0.0015	--	--
3 - Mile	Pelagic	kg/day	0.0073	--	--
	Benthic	kg/day	0.0018	--	--
5.5 - Mile	Pelagic	kg/day	0.0092	--	--
	Benthic	kg/day	0.0023	--	--
12.7 - Mile	Pelagic	kg/day	0.0183	--	--
	Benthic	kg/day	0.0046	--	--

Skin Surface Area (surface water) Location Specific Values				
		Values		
Location	units	Adult (18+)	Youth (7 to 18)	Child (1 to 6)
South Spring	cm^2	1225	949	451
North Spring	cm^2	1225	949	451
Pre-A	cm^2	1521	1352	776
A	cm^2	1818	1754	939
B	cm^2	2410	2157	1101
C	cm^2	3595	2559	1101
D	cm^2	3595	2559	1101
E	cm^2	3595	2559	1101
F	cm^2	3595	2559	2284
G	cm^2	--	--	--
1 - Mile	cm^2	3002.5	2358	1101
3 - Mile	cm^2	3595	2559	1101

TABLE A-1

REVISED GENERAL AND CHEMICAL-SPECIFIC EXPOSURE PARAMETER VALUES
HUMAN HEALTH RISK ASSESSMENT
NEAL'S LANDFILL
BLOOMINGTON, MONROE COUNTY, INDIANA

Notes:

ABS	=	Dermal absorption factor (unitless)
Cw	=	Chemical concentration in water - this concentration is equivalent to the location-specific surface water exposure point concentration (EPA) (see Table A-2) in units of milligrams per cubic meter (mg/cm^3).
FA	=	Fraction absorbed water (unitless)
GI	=	Gastrointestinal absorption (unitless)
K_p	=	Dermal permeability coefficient in water (centimeter per hour [cm/hr])
PCB	=	Polychlorinated biphenyl
RfD	=	Oral reference dose (milligram per kilogram - day [mg/kg-day])
SF	=	Oral slope factor $(mg/kg\text{-}day)^{-1}$
τ_{event}	=	Lag time per event (hour per event [hr/event])
t_{event}	=	Event duration (hour per event [hr/event])
TEQ	=	Toxicity equivalent

TABLE A-2

MEDIUM-SPECIFIC EXPOSURE POINT CONCENTRATIONS
HUMAN HEALTH RISK ASSESSMENT
NEAL'S LANDFILL
BLOOMINGTON, MONROE COUNTY, INDIANA

Exposure Point Concentrations[a]								
		Values						
Location[b]	Chemical	$EPC_{al(pelagic)}$ (ug/kg-PCB/ng/kg-TEQ)	$EPC_{al(benthic)}$ (ug/kg-PCB/ng/kg-TEQ)	EPC_{sw} (mg/L)	DA_{event} (mg/cm^2-event)	EPC_{sed} (mg/kg)	EPC_{bs} (mg/kg)	EPC_{fs} (mg/kg)
South Spring	PCB	--	--	1.25E-03	3.53E-06	--	--	--
North Spring	PCB	--	--	6.76E-04	1.91E-06	--	--	--
Pre-A	PCB	--	--	9.28E-04	2.62E-06	--	12.5	4.5
A	PCB	--	--	1.91E-04	5.39E-07	1	1.8	2.7
B	PCB	--	--	--	--	3.20E-02	--	2.20E-01
C	PCB	--	--	--	--	--	1.40E-02	--
D	PCB	--	--	--	--	1.90E-02	--	--
E	PCB	--	--	--	--	2.70E-02	2.40E-01	--
F	PCB	--	--	--	--	3.60E-01	--	--
G	PCB	--	--	--	--	2.10E-01	--	--
1 - Mile	PCB	182	808	1.04E-04	2.94E-07	--	--	--
	TEQ	3.0	10.2	--	--	--	--	--
3 - Mile	PCB	114.8	182.8	1.50E-04	4.24E-07	--	--	--
	TEQ	1.0	1.4	--	--	--	--	--
5.5 - Mile	PCB	--	192	--	--	--	--	--
	TEQ	--	2.3	--	--	--	--	--
12.7 - Mile	PCB	92	165	--	--	--	--	--
	TEQ	--	--	--	--	--	--	--

Notes:

--	=	Not applicable
ug/kg	=	Microgram per kilogram
mg/kg	=	Milligram per kilogram
mg/L	=	Milligram per liter
ng/kg	=	Nanogram per kilogram
PCB	=	Polychlorinated biphenyl
TEQ	=	Toxicity equivalent

a Medium-specific EPC calculations are discussed in Section 3.3.1 and presented in Appendix A of the draft HHRA (Tetra Tech 2006).

b See Figures 1, 2, and 3 in the draft HHRA (Tetra Tech 2006).

TABLE [illegible]-3

REVISED FISH INGESTION RISKS AND HAZARDS
HUMAN HEALTH RISK ASSESSMENT
NEAL'S LANDFILL
BLOOMINGTON, MONROE COUNTY, INDIANA

Location	Fish	PCB				TCDD TEQ				
		ADD	Hazard	LADD	Risk	ADD	Hazard	LADD	Risk	$Risk_{EPA\ Alt}$
Adult (18 or more years old)										
1 - Mile	Pelagic	7.8E-06	3.9E-01	3.3E-06	6.7E-06	1.3E-10	1.3E-01	5.5E-11	8.3E-06	5.5E-05
	Benthic	8.7E-06	4.3E-01	3.7E-06	7.4E-06	1.1E-10	1.1E-01	4.7E-11	7.0E-06	4.7E-05
3 - Mile	Pelagic	6.0E-06	3.0E-01	2.6E-06	5.1E-06	5.2E-11	5.2E-02	2.2E-11	3.4E-06	2.2E-05
	Benthic	2.4E-06	1.2E-01	1.0E-06	2.0E-06	1.8E-11	1.8E-02	7.7E-12	1.2E-06	7.7E-06
5.5 - Mile	Pelagic	--	--	--	--	--	--	--	--	--
	Benthic	3.2E-06	1.6E-01	1.4E-06	2.7E-06	3.8E-11	3.8E-02	1.6E-11	2.4E-06	1.6E-05
12.7 - Mile	Pelagic	1.2E-05	6.0E-01	5.2E-06	1.0E-05	--	--	--	--	--
	Benthic	5.4E-06	2.7E-01	2.3E-06	4.6E-06	--	--	--	--	--

Notes:

--	= Not calculated
ADD	= Average daily dose (mg/kg-day)
LADD	= Lifetime average daily dose (mg/kg-day)
mg/kg-day	= Milligram per kilogram per day
PCB	= Polychlorinated biphenyl
TCDD	= Tetrachloro-p-dioxin
TEQ	= Toxicity equivalent
$Risk_{EPA\ Alt}$	= U.S. Environmental Protection Agency's alternative TCDD slope factor (see Section 4.2.2).

TABLE A-4

**REVISED SURFACE WATER RISKS AND HAZARDS
HUMAN HEALTH RISK ASSESSMENT
NEAL'S LANDFILL
BLOOMINGTON, MONROE COUNTY, INDIANA**

Location	Chemical	Ingestion				Dermal Contact			
		ADD	Hazard	LADD	Risk	ADD	Hazard	LADD	Risk
Adult (18 or more years old)									
South Spring	PCB	7.3E-09	3.7E-04	3.1E-09	6.3E-09	3.4E-06	1.7E-01	1.5E-06	2.9E-06
North Spring	PCB	4.0E-09	2.0E-04	1.7E-09	3.4E-09	1.8E-06	9.2E-02	7.8E-07	1.6E-06
Pre-A	PCB	5.4E-09	2.7E-04	2.3E-09	4.7E-09	3.1E-06	1.6E-01	1.3E-06	2.7E-06
A	PCB	1.1E-09	5.6E-05	4.8E-10	9.6E-10	7.7E-07	3.8E-02	3.3E-07	6.6E-07
B	PCB	--	--	--	--	--	--	--	--
C	PCB	--	--	--	--	--	--	--	--
D	PCB	--	--	--	--	--	--	--	--
E	PCB	--	--	--	--	--	--	--	--
F	PCB	--	--	--	--	--	--	--	--
G	PCB	--	--	--	--	--	--	--	--
1 - Mile	PCB	2.1E-09	1.0E-04	8.9E-10	1.8E-09	2.3E-06	1.2E-01	1.0E-06	2.0E-06
3 - Mile	PCB	1.3E-09	6.6E-05	5.7E-10	1.1E-09	1.8E-06	8.9E-02	7.7E-07	1.5E-06
Youth (7 to 18 years old)									
South Spring	PCB	2.2E-08	1.1E-03	3.7E-09	7.5E-09	3.9E-06	2.0E-01	6.7E-07	1.3E-06
North Spring	PCB	1.2E-08	5.9E-04	2.0E-09	4.1E-09	2.1E-06	1.1E-01	3.6E-07	7.2E-07
Pre-A	PCB	1.6E-08	8.1E-04	2.8E-09	5.6E-09	4.1E-06	2.1E-01	7.1E-07	1.4E-06
A	PCB	3.3E-09	1.7E-04	5.7E-10	1.1E-09	1.1E-06	5.5E-02	1.9E-07	3.8E-07
B	PCB	--	--	--	--	--	--	--	--
C	PCB	--	--	--	--	--	--	--	--
D	PCB	--	--	--	--	--	--	--	--
E	PCB	--	--	--	--	--	--	--	--
F	PCB	--	--	--	--	--	--	--	--
G	PCB	--	--	--	--	--	--	--	--
1 - Mile	PCB	2.7E-09	1.4E-04	4.7E-10	9.4E-10	1.2E-06	6.1E-02	2.1E-07	4.2E-07
3 - Mile	PCB	3.9E-09	2.0E-04	6.7E-10	1.3E-09	1.9E-06	9.5E-02	3.2E-07	6.5E-07
Child (1 to 6 years old)									
South Spring	PCB	6.8E-08	3.4E-03	5.9E-09	1.2E-08	5.8E-06	2.9E-01	5.0E-07	1.0E-06
North Spring	PCB	3.7E-08	1.9E-03	3.2E-09	6.3E-09	3.1E-06	1.6E-01	2.7E-07	5.4E-07
Pre-A	PCB	5.1E-08	2.5E-03	4.4E-09	8.7E-09	7.4E-06	3.7E-01	6.4E-07	1.3E-06
A	PCB	1.0E-08	5.2E-04	9.0E-10	1.8E-09	1.9E-06	9.3E-02	1.6E-07	3.2E-07
B	PCB	--	--	--	--	--	--	--	--
C	PCB	--	--	--	--	--	--	--	--
D	PCB	--	--	--	--	--	--	--	--
E	PCB	--	--	--	--	--	--	--	--
F	PCB	--	--	--	--	--	--	--	--
G	PCB	--	--	--	--	--	--	--	--
1 - Mile	PCB	8.5E-09	4.3E-04	7.3E-10	1.5E-09	1.8E-06	8.9E-02	1.5E-07	3.0E-07
3 - Mile	PCB	1.2E-08	6.2E-04	1.1E-09	2.1E-09	2.6E-06	1.3E-01	2.2E-07	4.4E-07

Notes:

--	= Not calculated
ADD	= Average daily dose (mg/kg-day)
LADD	= Lifetime average daily dose (mg/kg-day)
mg/kg-day	= Milligram per kilogram per day
PCB	= Polychlorinated biphenyl

TABLE A-5

REVISED SEDIMENT AND SOIL EXPOSURES, RISKS, AND HAZARDS
HUMAN HEALTH RISK ASSESSMENT
NEAL'S LANDFILL
BLOOMINGTON, MONROE COUNTY, INDIANA

Location	Type	Chemical	Ingestion				Dermal Contact			
			ADD	Hazard	LADD	Risk	ADD	Hazard	LADD	Risk
Adult (18 or more years old)										
Pre-A	Bank Soil	PCB	9.8E-07	4.9E-02	4.2E-07	8.4E-07	5.5E-07	2.7E-02	2.3E-07	4.7E-07
	Floodplain Soil	PCB	3.5E-07	1.8E-02	1.5E-07	3.0E-07	2.0E-07	9.8E-03	8.4E-08	1.7E-07
A	Sediment	PCB	7.8E-08	3.9E-03	3.4E-08	6.7E-08	7.0E-08	3.5E-03	3.0E-08	6.0E-08
	Bank Soil	PCB	1.4E-07	7.0E-03	6.0E-08	1.2E-07	7.9E-08	3.9E-03	3.4E-08	6.7E-08
	Floodplain Soil	PCB	2.1E-07	1.1E-02	9.1E-08	1.8E-07	1.2E-07	5.9E-03	5.1E-08	1.0E-07
B	Sediment	PCB	8.5E-09	4.3E-04	3.6E-09	7.3E-09	7.6E-09	3.8E-04	3.3E-09	6.5E-09
	Floodplain Soil	PCB	6.5E-08	3.3E-03	2.8E-08	5.6E-08	3.7E-08	1.8E-03	1.6E-08	3.1E-08
C	Sediment	PCB	--	--	--	--	--	--	--	--
	Bank Soil	PCB	4.2E-09	2.1E-04	1.8E-09	3.6E-09	2.3E-09	1.2E-04	1.0E-09	2.0E-09
D	Sediment	PCB	5.1E-09	2.5E-04	2.2E-09	4.3E-09	4.5E-09	2.3E-04	1.9E-09	3.9E-09
	Bank Soil	PCB	--	--	--	--	--	--	--	--
E	Sediment	PCB	3.2E-09	1.6E-04	1.4E-09	2.7E-09	2.8E-09	1.4E-04	1.2E-09	2.4E-09
	Bank Soil	PCB	2.8E-08	1.4E-03	1.2E-08	2.4E-08	1.6E-08	7.9E-04	6.7E-09	1.3E-08
F	Sediment	PCB	4.2E-08	2.1E-03	1.8E-08	3.6E-08	3.8E-08	1.9E-03	1.6E-08	3.2E-08
	Bank Soil	PCB	--	--	--	--	--	--	--	--
G	Sediment	PCB	2.5E-08	1.2E-03	1.1E-08	2.1E-08	2.2E-08	1.1E-03	9.4E-09	1.9E-08
	Bank Soil	PCB	--	--	--	--	--	--	--	--
Youth (7 to 18 years old)										
Pre-A	Bank Soil	PCB	1.5E-06	7.3E-02	2.5E-07	5.0E-07	1.8E-06	8.9E-02	3.1E-07	6.1E-07
	Floodplain Soil	PCB	5.2E-07	2.6E-02	9.0E-08	1.8E-07	6.4E-07	3.2E-02	1.1E-07	2.2E-07
A	Sediment	PCB	1.2E-07	5.8E-03	2.0E-08	4.0E-08	8.0E-08	4.0E-03	1.4E-08	2.8E-08
	Bank Soil	PCB	2.1E-07	1.0E-02	3.6E-08	7.2E-08	2.6E-07	1.3E-02	4.4E-08	8.8E-08
	Floodplain Soil	PCB	3.1E-07	1.6E-02	5.4E-08	1.1E-07	3.9E-07	1.9E-02	6.6E-08	1.3E-07
B	Sediment	PCB	1.3E-08	6.3E-04	2.2E-09	4.3E-09	8.7E-09	4.4E-04	1.5E-09	3.0E-09
	Floodplain Soil	PCB	9.7E-08	4.9E-03	1.7E-08	3.3E-08	1.2E-07	6.0E-03	2.0E-08	4.1E-08
C	Sediment	PCB	--	--	--	--	--	--	--	--
	Bank Soil	PCB	6.2E-09	3.1E-04	1.1E-09	2.1E-09	7.6E-09	3.8E-04	1.3E-09	2.6E-09
D	Sediment	PCB	7.5E-09	3.8E-04	1.3E-09	2.6E-09	5.2E-09	2.6E-04	8.9E-10	1.8E-09
	Bank Soil	PCB	--	--	--	--	--	--	--	--
E	Sediment	PCB	4.7E-09	2.4E-04	8.1E-10	1.6E-09	3.3E-09	1.6E-04	5.6E-10	1.1E-09
	Bank Soil	PCB	4.2E-08	2.1E-03	7.2E-09	1.4E-08	5.1E-08	2.6E-03	8.8E-09	1.8E-08
F	Sediment	PCB	0.0E+00	0.0E+00	1.1E-08	2.2E-08	4.3E-08	2.2E-03	7.4E-09	1.5E-08
	Bank Soil	PCB	--	--	--	--	--	--	--	--
G	Sediment	PCB	3.7E-08	1.8E-03	6.3E-09	1.3E-08	2.5E-08	1.3E-03	4.3E-09	8.7E-09
	Bank Soil	PCB	--	--	--	--	--	--	--	--

TABLE A-5

REVISED SEDIMENT AND SOIL EXPOSURES, RISKS, AND HAZARDS
HUMAN HEALTH RISK ASSESSMENT
NEAL'S LANDFILL
BLOOMINGTON, MONROE COUNTY, INDIANA

Location	Type	Chemical	Ingestion				Dermal Contact			
			ADD	Hazard	LADD	Risk	ADD	Hazard	LADD	Risk
Child (1 to 6 years old)										
Pre-A	Bank Soil	PCB	9.1E-06	4.6E-01	7.8E-07	1.6E-06	3.0E-06	1.5E-01	2.6E-07	5.2E-07
	Floodplain Soil	PCB	3.3E-06	1.6E-01	2.8E-07	5.6E-07	1.1E-06	5.5E-02	9.4E-08	1.9E-07
A	Sediment	PCB	7.3E-07	3.7E-02	6.3E-08	1.3E-07	1.2E-07	6.2E-03	1.1E-08	2.1E-08
	Bank Soil	PCB	1.3E-06	6.6E-02	1.1E-07	2.3E-07	4.4E-07	2.2E-02	3.8E-08	7.5E-08
	Floodplain Soil	PCB	2.0E-06	9.9E-02	1.7E-07	3.4E-07	6.6E-07	3.3E-02	5.6E-08	1.1E-07
B	Sediment	PCB	7.9E-08	4.0E-03	6.8E-09	1.4E-08	1.4E-08	6.8E-04	1.2E-09	2.3E-09
	Floodplain Soil	PCB	6.1E-07	3.1E-02	5.2E-08	1.0E-07	2.0E-07	1.0E-02	1.7E-08	3.5E-08
C	Sediment	PCB	--	--	--	--	--	--	--	--
	Bank Soil	PCB	3.9E-08	1.9E-03	3.3E-09	6.7E-09	1.3E-08	6.5E-04	1.1E-09	2.2E-09
D	Sediment	PCB	4.7E-08	2.4E-03	4.0E-09	8.1E-09	8.0E-09	4.0E-04	6.9E-10	1.4E-09
	Bank Soil	PCB	--	--	--	--	--	--	--	--
E	Sediment	PCB	3.0E-08	1.5E-03	2.5E-09	5.1E-09	5.0E-09	2.5E-04	4.3E-10	8.6E-10
	Bank Soil	PCB	2.6E-07	1.3E-02	2.3E-08	4.5E-08	8.8E-08	4.4E-03	7.5E-09	1.5E-08
F	Sediment	PCB	3.9E-07	2.0E-02	3.4E-08	6.8E-08	6.7E-08	3.4E-03	5.7E-09	1.1E-08
	Bank Soil	PCB	--	--	--	--	--	--	--	--
G	Sediment	PCB	2.3E-07	1.2E-02	2.0E-08	3.9E-08	3.9E-08	2.0E-03	3.4E-09	6.7E-09
	Bank Soil	PCB	--	--	--	--	--	--	--	--

Notes:

-- = Not calculated
ADD = Average daily dose (mg/kg-day)
LADD = Lifetime average daily dose (mg/kg-day)
mg/kg-day = Milligram per kilogram per day
PCB = Polychlorinated biphenyl

ATTACHMENT 3

Age structure and growth of *Semotilus atromaculatus* (Mitchill) in PCB-contaminated streams

Journal of Fish Biology (2006) **68**, 44–62
doi:10.1111/j.1095-8649.2005.00875.x, available online at http://www.blackwell-synergy.com

Age structure and growth of *Semotilus atromaculatus* (Mitchill) in PCB-contaminated streams

D. S. HENSHEL*†, D. W. SPARKS‡, T. P. SIMON‡ AND M. J. TOSICK‡

**Indiana University, School of Public and Environmental Affairs, Bloomington, Indiana 47405, U.S.A. and ‡U.S. Fish and Wildlife Service, 620 S. Walker Street, Bloomington, Indiana 47403, U.S.A.*

(Received 14 September 2004, Accepted 15 June 2005)

Creek chub *Semotilus atromaculatus* from two PCB contaminated streams (Clear Creek and Richland Creek) at three locations and a reference stream (Little Indian Creek), Indiana, U.S.A., were examined to determine if age class structure and growth variables were correlated with *in-situ* PCB exposure. Approximately five to 15 fish were captured weekly during the spring spawning season and monthly thereafter for a 12 month period. Fish collected ranged from 25 to 267 mm total length (L_T). Throughout the course of this study, no spawning activity was observed at either location in Clear Creek, although some very small young-of-the-year (YOY) creek chub fry were observed at the downstream location by late summer. Creek chub nests were observed in both Richland Creek and Little Indian Creek but YOY were common only in Little Indian Creek. Exposure to PCBs was shown to both enhance and decrease growth in varied laboratory tests; subtle but significant gender-specific differences in the growth of creek chub populations between the sites were observed. Creek chub up to 24 months in age from Clear Creek and Richland Creek were significantly larger (both L_T and mass for males and L_T for females) than reference site creek chub. This trend was reversed for creek chub aged $\geq$24 months as the reference site fish were consistently larger with reference males weighing significantly more. Older age classes of creek chub were missing in areas of higher PCB contamination. Female population growth rates and individual instantaneous growth rates were consistently higher at the reference site in comparison to the PCB-contaminated sites. Calculation of 'functional b' (as a condition factor) did indicate that growth enhancement in young males did occur at the most contaminated site and reductions in growth (mass relative to L_T) occurred in females from all contaminated sites. Furthermore, long-term survivorship for females was reduced in the PCB-contaminated streams. All of these subtle alterations in growth would not have been observed if males and females had not been analysed separately.

Key words: age structure; growth; PCBs; *Semotilus atromaculatus*; sex differences.

INTRODUCTION

The first decade of laboratory research on the effects of commercial mixtures of polychlorinated biphenyls (PCBs) on aquatic organisms focused on classical toxicity testing (Hansen *et al.*, 1974*a*, *b*, 1975; Nebeker & Puglisi, 1974;

†Author to whom correspondence should be addressed. Tel.: +1 812 855 4672; fax: +1 812 855 7802; email: dhenshel@indiana.edu

Nebeker *et al.*, 1974; Schimmel *et al.*, 1974; DeFoe *et al.*, 1978). As part of these studies, both growth enhancement and growth reduction were often observed (Nebeker *et al.*, 1974; Hansen *et al.*, 1976; DeFoe *et al.*, 1978; Mauck *et al.*, 1978; Bengtsson, 1980; Mac & Seelye, 1981*a*, *b*). Growth reduction has been typically identified as 'wasting away' leading to death (Halter & Johnson, 1974; Bengtsson, 1980; Cleland *et al.*, 1988). Bengtsson (1979) stated that artificially stimulated growth is not a benefit, rather it is an adverse effect. Although he did not understand the underlying mechanism of this growth enhancement, Bengtsson (1979) noted that thyroid hormones regulate fish growth. Much research has documented that PCBs alter thyroid hormone (T) levels in a variety of fish species (Mayer *et al.*, 1977; Leatherland *et al.*, 1978; Fingerman & Russell, 1980; Besselink *et al.*, 1996; Palace *et al.*, 2001; Brown *et al.*, 2004) although the pattern of whether T_3 or T_4 is more affected, and the dynamic changes in response to dose and over time seems to be complex and variable by species.

Great strides have been made in understanding the sublethal effects of PCBs in the past 25 years including the relationship between the toxicity of dioxin (Walker *et al.*, 1991, 1994; Wright & Tillitt, 1999) and the dioxin-like properties of some congeners of PCBs (Ankley *et al.*, 1991; Van den Berg *et al.*, 1998). The main focus of this research effort, however, has been on reproductive success and the mode of toxicity of dioxin and dioxin-like PCBs (Cantrell *et al.*, 1998*a*, *b*; Thomas, 1999). Perhaps this is due in part to Niimi (1992) who concluded that there are no acutely toxic problems associated with environmental concentrations of PCBs because fishes do not accumulate lethal concentrations of PCBs anymore. While this generalization may be accurate for most of North America, given the lack of in-depth field studies present in the literature, this position is questionable. More recently, Black *et al.* (1998*a*, *b*) not only documented reproductive effects in mummichog *Fundulus heteroclitus* (L.) from the New Bedford Harbor Superfund site (MA, U.S.A.), but also documented reduced survivorship in older females which correlated with total PCB concentrations. Subtle but serious effects can still be found in field situations and in the laboratory at environmentally relevant PCB concentrations (Monosson *et al.*, 1994). A comprehensive investigation of creek chub *Semotilus atromaculatus* (Mitchell) was initiated because it is common throughout eastern North America and is known to be present in many streams in Indiana that are highly contaminated with PCBs (IDEM, 2004). The objective of the study seek was to determine if age class structure and growth endpoints are sensitive indicators of adverse impacts relative to ambient PCB concentrations.

MATERIALS AND METHODS

SITE DESCRIPTION

Lemon Lane Landfill (LL) and Neal's Landfill (NL) are National Priorities List (Superfund) sites located in Monroe County, Indiana, U.S.A. Creek chub were collected from Clear Creek (CC) at two locations downstream of the primary discharge of PCB-contaminated groundwater emanating from LL. Clear Creek 1

(CC1; drainage area 5·96 km^2; 39° 10′ N; 86° 33′ W) was located 1400 m downstream and CC2 (drainage area 16·32 km^2; 39° 9′ N; 86° 32′ W) was located 3100 m downstream of the LL discharge springs. Conard's Branch and Richland Creek (CB) (drainage area 8·03 km^2; 39° 11′ N; 86° 39′ W) were also sampled *c.* 800 m downstream of the primary discharge of PCB-contaminated groundwater emanating from NL. The reference site for the study was on Little Indian Creek (LI; 39°0 N; 86° 41′ W) just upstream of its confluence with Indian Creek (drainage area 27·45 km^2). All four sites are in the White River drainage, Monroe County, Indiana, U.S.A. (Fig. 1). Some qualitative habitat characteristics for these sites are given in Table I. Over the course of many years, hundreds of samples have been collected from these study sites and chemically analysed for hazardous substances, including PCBs. Sample types included: groundwater at the source spring, ambient surface water at various downstream locations, sediment and whole creek chub (Table II). It was clear from the data collected that PCBs were the only significant bioaccumulating contaminant of concern at CC1, CC2 and CB study sites (ISDH, 1994). Only very low concentrations of PCBs are found in fish at LI, confirming that this site is suitable to represent background conditions for south-western Indiana.

EXPERIMENTAL DESIGN AND FIELD COLLECTION

Monthly collection, of five to 15 individual creek chub were conducted between April 1999 and April 2000 at all sites, usually within 24 h, during daylight hours using a battery back-pack pulsed-DC electrofishing unit capable of 2–3 A output. Fish were collected weekly during reproductive time (April to June) periods and semi-monthly during non-reproductive periods (July to March). Fish were collected and placed into a live well until the conclusion of the sampling zone. Creek chub were preserved in 10% formalin for future laboratory work.

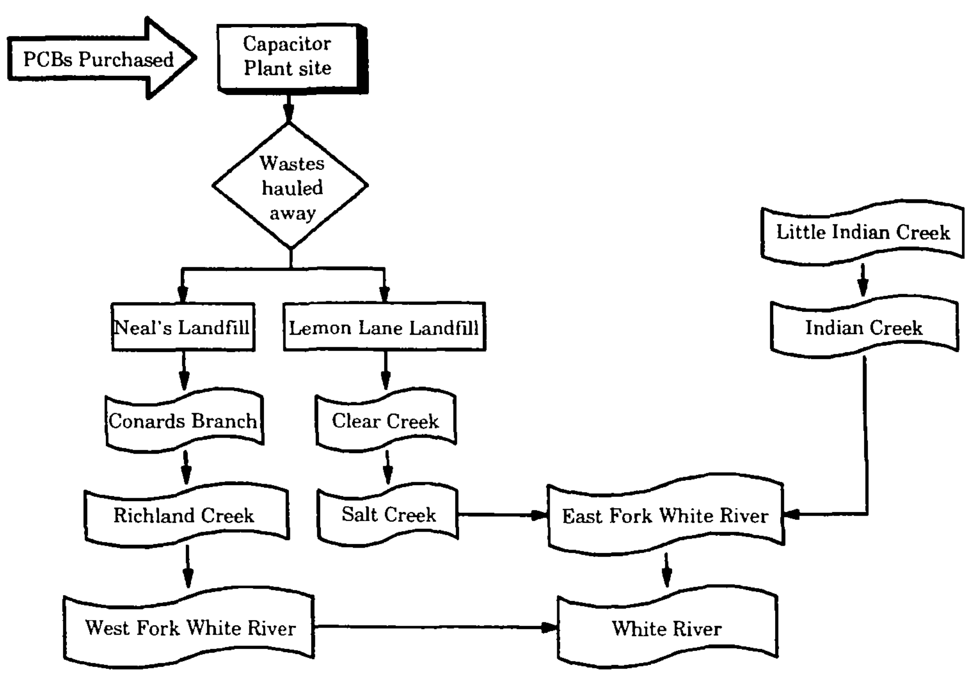

FIG. 1. Schematic of PCB study sites, Monroe County, IN, U.S.A.

TABLE I. Habitat characteristics of the study sites

	CC1	CB	CC2	LI
Source of PCBs	Lemon Lane LF	Neal's LF	Lemon Lane LF	Reference site
Drainage area (ha)	600	800	1600	2700
Substratum types (%)	Sand (40), gravel (35), cobble (20), boulder (5)	Silt (25), gravel (25), cobble (20), bedrock (30)	Sand (35), gravel (30), cobble (15), boulder (15)	Sand (5), gravel (20), cobble (60), boulder (5), bedrock (10)
Land use	Urban and residential	Agricultural	Urban and industrial	Agricultural
Habitat	Forested corridor	Forested corridor	Pasture and forested corridor	Forested corridor
Stream width (m)	2	4	3	6·5

CC1, upper Clear Creek; CB, Conards Branch and Richland Creek; CC2, lower Clear Creek; LI, Little Indian Creek.

AGE AND GROWTH METHODOLOGY

Preserved creek chub were rinsed in tap water, blotted dry, and whole masses (M_B) were measured to the nearest 0·1 µg. Total length (L_T) was measured to the nearest 0·1 mm from the anterior most part of the fish to the longest caudal fin ray length by depressing the caudal lobe dorso-ventrally (Anderson & Gutreuter, 1989). A scale sample was obtained above the lateral line, anterior to the base of the dorsal fin for ageing by counting annuli (Jearld, 1989) using a Leica MZ12·5 stereomicroscope equipped with an ocular micrometer and transmitted light using dark field lighting. The age of 586 fish from all sites, including age 0 year fish, was determined. The Dahl-Lea method (a direct proportion basis) was used to determine backcalculated L_T-at-age (Lagler, 1956). A total of 474 creek chub were $\geq$ age 1 years and were included in the backcalculation of size at successive annuli (Ricker, 1975). Ordinary and functional b (slope, condition factor) regressions were calculated (Ricker, 1975) according to the following formulae: $M_B = aL_T^b$ or $\log_{10} M_B = \log_{10} a + b \log_{10} L_T$, where M_B is in g, L_T in cm and $a =$ the intercept. Ricker (1975) states that 'usually the best available estimate of the growth rate of individual fish (G) comes from the back-calculation of their length at the last two annuli on the scales.' Individual instantaneous growth rate of length increase was

TABLE II. PCB concentrations (ranges and means ± S.D.) in various media at the study sites

	CC1	CB	CC2	LI
PCB groundwater concentrations at source (ppb)	5–470 (Tetra Tech, 2001)	0·1–8·7 (Tetra Tech, 2000)	5–470 (Tetra Tech, 2001)	NA
Ambient water at study area (ppb)	1·1 (USEPA, 1992)	0·46 (CBS, 1998)	<1·0 (USEPA, 1992)	NA
Sediments (ppm)	1·0 (USEPA, 1992) 4·3 (Normandeau, 1995) 2·17 ± 0·26, $n = 3$ (Westinghouse, 1997)	1·87 ± 0·93, $n = 5$ (CBS, 1998)	0·29 (USEPA, 1992) 0·58 (Normandeau, 1995) 0·56 ± 0·14, $n = 5$ (Westinghouse, 1997)	NA
Creek chub (whole body, ppm)	19·2 ± 3·2, $n = 9$ (Westinghouse, 1997)	12·1 ± 3·1, $n = 8$ (CBS, 1998)	2·1 ± 0·2, $n = 9$ (Westinghouse, 1997)	0·01 ± 0·00, $n = 6$ (USFWS, unpubl. data)

CC1, upper Clear Creek; CB, Conards Branch and Richland Creek; CC2, lower Clear Creek; LI, Little Indian Creek; ppb, parts per billion; NA, not analysed; ppm, parts per million; n, number of samples analysed.

calculated for each age class interval available at the study sites (Ricker, 1975) according to the following formula: $G = b(\ln L_{T_2} - \ln L_{T_1})$, where G = individual instantaneous growth rate of L_T increase, b = is the calculated functional b for that age class, L_{T_1} = the backcalculated L_T of the fish at the second to last annuli, and L_{T_2} = the backcalculated L_T of the fish at its last annuli. Population instantaneous growth rate of L_T increase (G_x) was obtained by comparing the mean size of surviving fish at successive ages in the same manner (Ricker, 1975). All calculations were initially conducted with males and females together, a common practice despite the fact that creek chub are sexually dimorphic. When it was determined that there were differences in survivorship between the sexes, all analyses were redone separately for both males and females.

STATISTICS

All statistical analyses were done using SAS (SAS Institute Inc, Cary, NC, U.S.A.). Total length and M_B were correlated to age (determined by year and by month) using PROC REG, a general purpose procedure for linear regression. PROC GLM (accepting an imbalanced design for the ANOVA) was used to evaluate significant differences between the sites. For the data determined by year, a variable for season collected (SEASON; April to June, spawning; July to September, postspawning; October to December, autumn; January to March, prespawning) was also included in the ANOVA. For this analysis 1 April of each year was assumed to be the estimated hatching date of creek chub. In addition to statistically analysing these data using age derived in years, age was also determined in months and the seasonality was then represented by Julian calendar date collected (JDATECOL). All graphs were created in Excel (Microsoft Corp.).

RESULTS

Equations for M_B and L_T regressed on age (in years) are presented for males and females for the study sites in Table III. The slope (b) of the mass and age equations for both males and females decreased with increasing mean PCB concentration in fish previously collected at these sites (Table III). The older age classes, especially females, were missing from the contaminated sites as evidenced by L_T-at-age and M_B-at-age figures (Fig. 2). For males, both L_T and M_B were significantly increased at the PCB sites for creek chub <24 months in age (L_T, $P < 0.001$–0.05; M_B, $P < 0.02$–0.04). Only L_T was significantly increased for females <24 months of age ($P < 0.001$–0.03). By age 2 years, there were no significant differences between the size of creek chub between these sites. By age 3 and 4 years, the male creek chubs from the reference site (LI) weighed significantly more than fish from the PCB-contaminated sites ($P < 0.001$–0.04). Age 3 years females also followed this trend, however these differences were not statistically significant (Fig. 2). At the reference site, males weighed significantly more than females in years 2 ($P = 0.06$), 3 ($P = 0.05$) and 4 ($P < 0.001$). There were no significant differences in M_B between males and females in any age class from the PCB contaminated sites.

Fish <40 mm in size (representing early age 0 year fish) were below expected frequencies in the PCB-contaminated streams (Fig. 3). This was not based on electrofishing gear bias because complete coverage was achieved in these streams during sampling. The sites were thoroughly examined for spawning activity. Creek chub successfully spawned at LI and CB, but not at CCl. Although

TABLE III. Regression equations for body mass (M_B, g) and total length (L_T, mm) with age (x, years) by site and sex

Site	Sex	M_B		L_T	
LI	F	$13·24\,x - 7·08$	($P < 0·001$, $r^2 = 0·74$)	$31·97\,x + 39·24$	($P < 0·001$, $r^2 = 0·82$)
CC1	F	$6·72\,x + 0·40$	($P < 0·001$, $r^2 = 0·65$)	$26·91\,x + 51·80$	($P < 0·001$, $r^2 = 0·69$)
CB	F	$9·42\,x - 2·93$	($P < 0·001$, $r^2 = 0·57$)	$36·81\,x + 36·44$	($P < 0·001$, $r^2 = 0·60$)
CC2	F	$10·68\,x - 2·94$	($P < 0·001$, $r^2 = 0·58$)	$34·59\,x + 44·42$	($P < 0·001$, $r^2 = 0·70$)
LI	M	$21·62\,x - 13·12$	($P < 0·001$, $r^2 = 0·52$)	$38·52\,x + 37·26$	($P < 0·001$, $r^2 = 0·80$)
CC1	M	$12·66\,x - 3·77$	($P < 0·001$, $r^2 = 0·54$)	$30·17\,x + 50·16$	($P < 0·001$, $r^2 = 0·62$)
CB	M	$14·19\,x - 5·18$	($P < 0·001$, $r^2 = 0·67$)	$36·46\,x + 39·41$	($P < 0·001$, $r^2 = 0·71$)
CC2	M	$21·44\,x - 14·26$	($P < 0·001$, $r^2 = 0·72$)	$35·43\,x + 46·35$	($P < 0·001$, $r^2 = 0·83$)

LI, Little Indian Creek; CC1, upper Clear Creek; CB, Conards Branch and Richland Creek; CC2, lower Clear Creek; F, female; M, male.

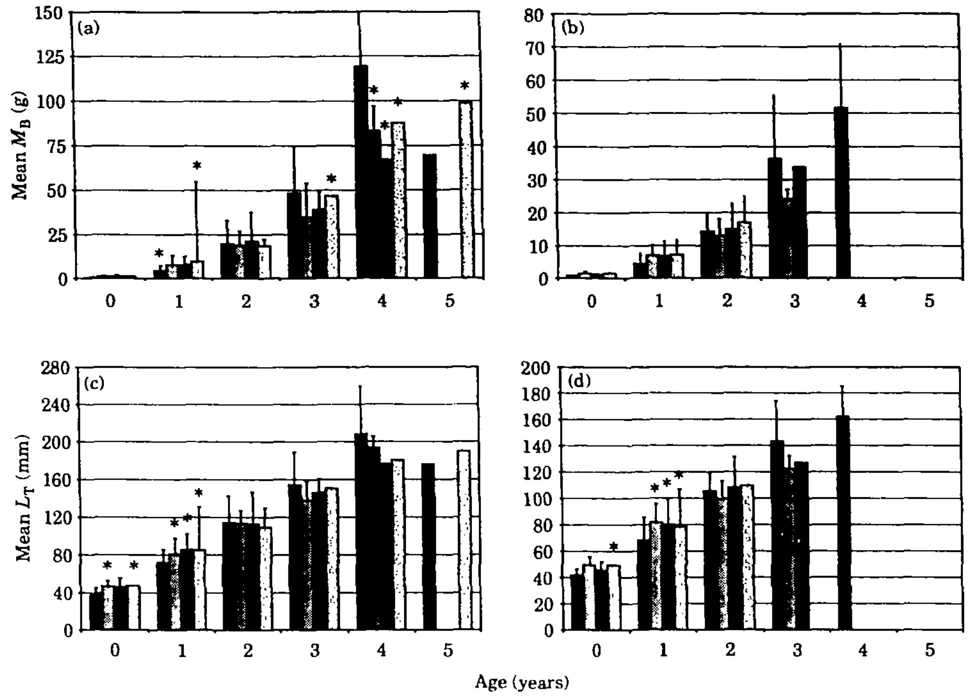

FIG. 2. Creek chub mean (a) male and (b) female masses, and (c) male and (d) female total length at Little Indian Creek (■), upper Clear Creek (□), Conards Branch and Richland Creek (▩) and lower Clear Creek (▨). *, significant differences from reference site (Little Indian Creek) at $P < 0.05$.

creek chub spawning was not specifically observed at CC2, juveniles were observed in this location by mid-summer. The reduced numbers of larger fish at the PCB contaminated sites are also indicated in Fig. 4.

Instantaneous growth rates and mean calculated L_T at successive annuli are given in Tables IV and V. Within all of the age classes of females, both population and mean individual G were higher for the reference site (LI) fish. For males, G did not appear to have clear trends.

Mass and L_T relationships are shown in Tables VI and VII. Based on functional b values, it appears that growth enhancement (*i.e.* high mass relative to L_T) was occurring in CC1 males, the most contaminated site studied. For females at CC1, the exact opposite was true (*i.e.* reduced mass relative to L_T). Reference females from LI exhibited an increased mass relative to L_T when compared to females from all other sites.

DISCUSSION

Carlander (1969) presented summaries of published creek chub data from many sites in North America and stated that creek chub grow most rapidly in the mid-west. Overall, the L_T and M_B of creek chub from these four study sites were similar to the calculated mean M_B for creek chub age classes from MN, IA, NY, AL, MD and OH (Carlander, 1969) although sexes were combined for this

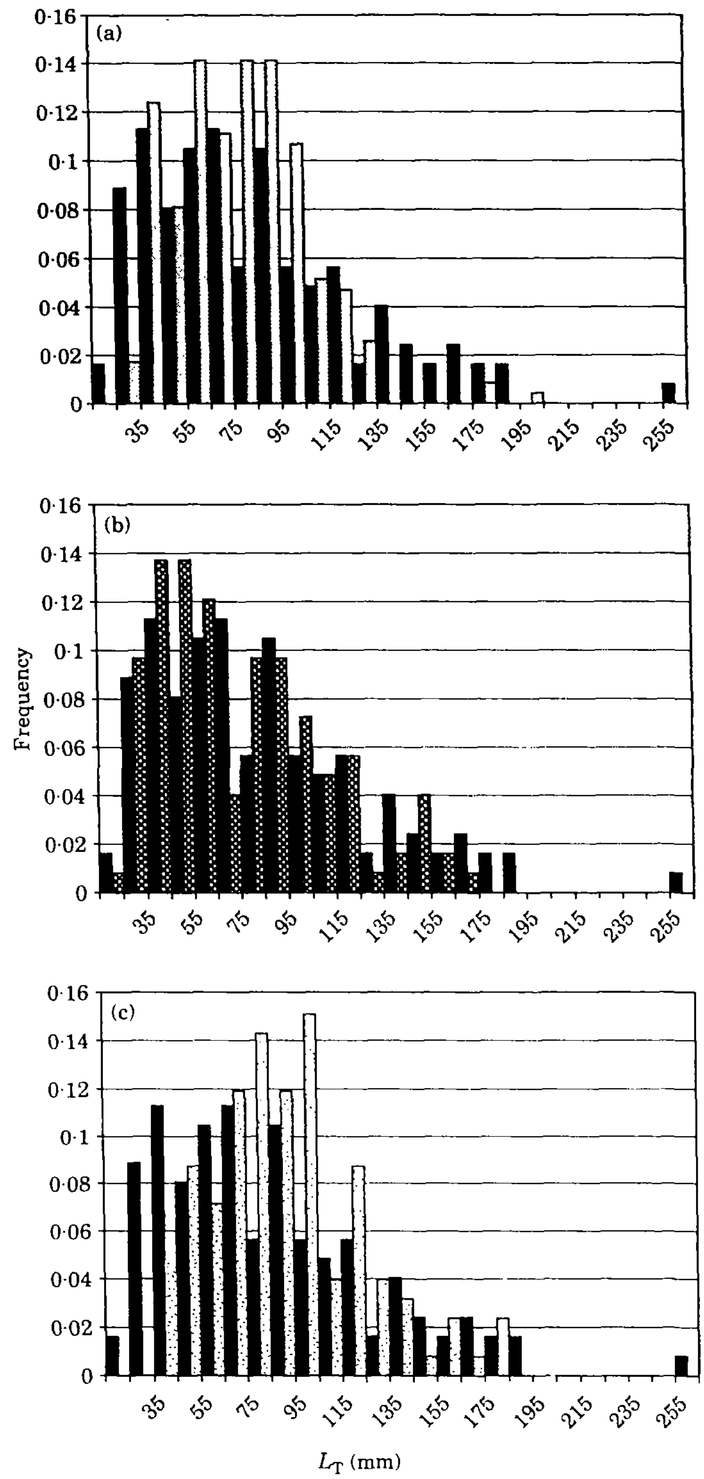

FIG. 3. Creek chub total length frequency comparisons for Little Indian Creek (■) and (a) upper Clear Creek (□), (b) Conards Branch and Richland Creek (▩) and (c) lower Clear Creek (▤).

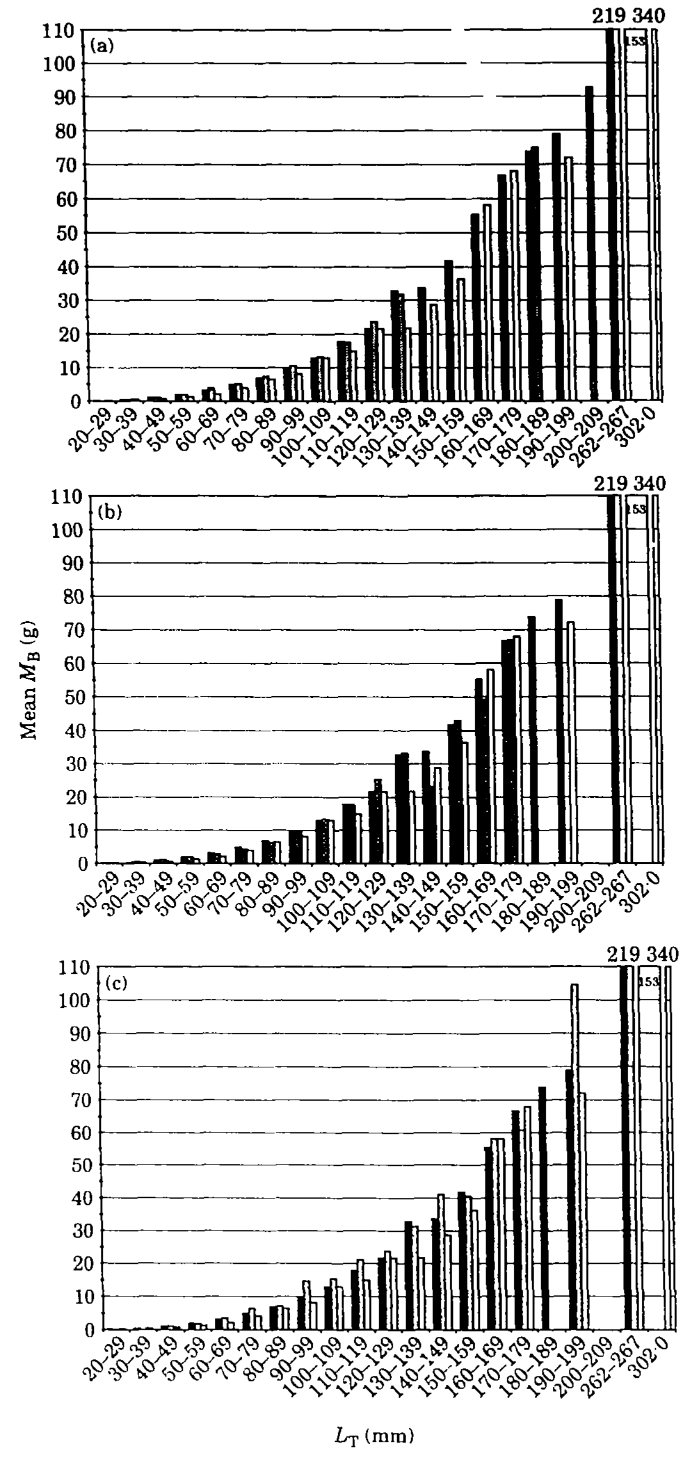

FIG. 4. Comparison of creek chub mean mass at 10 mm total length intervals between a North American mean (Carlander, 1969) (□), Little Indian Creek (■) and (a) upper Clear Creek (□), (b) Conards Branch and Richland Creek (▩) and (c) lower Clear Creek (▦).

TABLE IV. Male mean calculated total length at successive annuli and instantaneous growth rates by age class

Site	Age (years)	n	Mean L_T at capture (mm)	Mean calculated L_T at successive annuli (mm) 1	2	3	4	5	Age interval (years)	Population growth: Length interval (mm)	G_X	Individual growth: Length interval (mm)	G
LI	0	13	37·3										
LI	1	20	72·3	52·0									
LI	2	15	118·4	62·5	107·5				1–2	52–107·5	2·20	62·5–107·5	1·64
LI	3	5	154·2	53·0	90·9	142·0			2–3	107·5–142	0·85	90·9–142	1·36
LI	4	3	208·0	56·7	105·3	152·2	195·0		3–4	142–195	0·95	152–195	0·75
LI	5	1	176·0	32·7	85·7	112·1	119·9	140·2					
CC1	0	16	46·9										
CC1	1	54	82·0	52·9									
CC1	2	18	113·7	46·5	85·3				1–2	52·9–85·3	1·53	46·5–85·3	1·71
CC1	3	7	138·0	38·9	81·3	121·1			2–3	85·3–121·1	1·20	81·3–121	1·09
CC1	4	2	193·5	59·3	101·6	155·7	193·5		3–4	121·1–193·5	1·54	155·7–193·5	0·71
CC1	5	0											
CB	0	23	46·4										
CB	1	17	85·8	56·5									
CB	2	15	113·3	51·7	92·7				1–2	56·5–92·7	1·48	51·7–92·7	1·79
CB	3	5	145·9	47·4	87·5	137·7			2–3	92·7–137·7	1·25	87·5–137·7	1·29
CB	4	1	176·0	34·9	58·7	103·1	142·7		3–4	137·7–142·7	0·09	103·1–142·7	1·02
CB	5	0											
CC2	0	6	47·8										
CC2	1	31	85·2	53·4									
CC2	2	21	109·6	43·2	89·1				1–2	53·4–89·1	1·59	43·2–89·1	2·25
CC2	3	12	150·6	46·6	94·5	136·9			2–3	89·1–136·9	1·30	94·5–136·9	1·12
CC2	4	2	180·5	37·6	64·3	104·6	142·6		3–4	136·9–142·6	0·12	104·6–142·6	0·89
CC2	5	1	190·0	34·8	72·8	96·6	142·5	169·4					

LI, Little Indian Creek; CC1, upper Clear Creek; CB, Conards Branch and Richland Creek; CC2, lower Clear Creek; n, number; L_T, total length; G_X, population instantaneous growth rate; G, individual instantaneous growth rate.

TABLE V. Female mean calculated total length at successive annuli and instantaneous growth rates by age class

Site	Age (years)	n	Mean L_T at capture (mm)	Mean calculated L_T at successive annuli 1 (mm)	2	3	4	Age interval (years)	Population growth: Length interval (mm)	G_X	Individual growth: Length interval (mm)	G
LI	0	8	42·1									
LI	1	33	68·3	49·6								
LI	2	13	105·2	46·8	93·7			1–2	49·6–93·7	1·97	46·8–93·7	2·15
LI	3	4	143·1	52·7	100·8	137·9		2–3	93·7–137·9	1·26	100·8–137·9	1·02
LI	4	7	161·9	44·9	80·7	113·9	141·6	3–4	137·9–141·6	0·09	113·9–141·6	0·71
CC1	0	22	49·8									
CC1	1	67	81·8	52·7								
CC1	2	27	99·7	46·6	84·0			1–2	52·7–84	1·33	46·6–84	1·68
CC1	3	3	122·3	32·7	77·6	105·0		2–3	84–105	0·64	77·6–105	0·87
CC1	4	0						3–4				
CB	0	20	45·6									
CB	1	29	80·0	56·0								
CB	2	12	108·1	53·6	96·5			1–2	56–96·5	1·69	53–94	1·82
CB	3	1	127·0	52·9	99·5	124·9		2–3	96·5–124·9	0·72	90–136	0·63
CB	4	0						3–4				
CC2	0	5	49·6									
CC2	1	32	78·5	51·1								
CC2	2	13	109·5	47·6	87·6			1–2	51·1–87·6	1·63	47·6–87·6	1·85
CC2	3	0						2–3				
CC2	4	0						3–4				

LI, Little Indian Creek; CC1, upper Clear Creek; CB, Conards Branch and Richland Creek; CC2, lower Clear Creek; n, number; L_T, total length; G_X, population instantaneous growth rate; G, individual instantaneous growth rate.

TABLE VI. Male mass and total length regressions by age class

Site	Age (years)	n	Means $\log_{10}$ L_T	Means $\log_{10}$ M_B	Regression slope (b) Ordinary	Regression slope (b) Functional	S.D.	Functional intercept $\log_{10} a$
LI	1	20	1·85	0·62	3·00	3·03	0·22	−5·00
LI	2	15	2·07	1·25	3·02	3·04	0·22	−5·04
LI	3	5	2·18	1·61	2·98	3·01	0·28	−4·95
LI	4	3	2·31	2·01	2·87	2·87	0·29	−4·61
LI	5	1	2·25	1·84				
LI all 1+ years unweighted			2·00	1·07	3·02	3·03	0·44	−5·00
LI all 1+ years weighted			2·13	1·46	3·05	3·06	0·49	−5·05
CC1	1	54	1·91	0·82	2·72	3·20	0·24	−5·28
CC1	2	18	2·05	1·25	3·26	3·43	0·15	−5·80
CC1	3	7	2·14	1·50	3·13	3·28	0·16	−5·51
CC1	4	2	2·29	1·92	2·68	2·68		−4·21
CC1	5	0						
CC1 all 1+ years unweighted			1·97	1·00	2·86	2·93	0·31	−4·96
CC1 all 1+ years weighted			2·10	1·37	2·88	2·94	0·43	−4·67
CB	1	17	1·93	0·83	2·97	3·00	0·22	−5·10
CB	2	15	2·04	1·17	3·14	3·16	0·36	−5·10
CB	3	5	2·16	1·58	2·66	2·66	0·10	−3·44
CB	4	1	2·25	1·83				
CB	5	0						
CB all 1+ years unweighted			2·08	1·33	3·10	3·12	0·35	−5·17
CB all 1+ years weighted			1·97	0·97	3·12	3·12	0·42	−5·17
CC2	1	31	1·92	0·89	2·90	3·11	0·27	−5·08
CC2	2	21	2·03	1·23	2·97	3·03	0·18	−4·93
CC2	3	12	2·17	1·64	2·81	2·87	0·13	−4·59
CC2	4	2	2·25	1·92	3·70	3·70		−6·43
CC2	5	1	2·28	1·99				
CC2 all 1+ years unweighted			2·01	1·18	2·95	3·01	0·33	−4·90
CC2 all 1+ years weighted			2·13	1·53	3·06	3·07	0·41	−5·01

LI, Little Indian Creek; CC1, upper Clear Creek; CB, Conards Branch and Richland Creek; CC2, lower Clear Creek; n, number of individuals; M_B, mass (g); L_T, total length (mm); S.D., from the predictive regression line of $\log_{10} M_B$ on $\log_{10} L_T$.

TABLE VII. Female mass and total length regressions by age class

Site	Age (years)	n	Means $\log_{10} L_T$	Means $\log_{10} M_B$	Regression slope (b) Ordinary	Regression slope (b) Functional	S.D.	Functional intercept $\log_{10} a$
LI	1	33	1·82	0·53	3·08	3·10	0·31	−5·12
LI	2	13	2·02	1·12	3·18	3·25	0·16	−5·44
LI	3	4	2·15	1·48	3·24	3·26	0·33	−5·53
LI	4	7	2·21	1·69	2·56	2·68	0·12	−4·22
LI all 1+ years unweighted			1·97	0·96	3·01	3·02	0·45	−4·97
LI all 1+ years weighted			2·05	1·20	2·97	2·97	0·49	−4·88
CC1	1	67	1·91	0·79	2·77	2·85	0·18	−4·64
CC1	2	27	1·99	1·08	2·75	2·88	0·15	−4·66
CC1	3	3	2·09	1·38	1·12	1·48	0·05	−1·70
CC1	4	0						
CC1 all 1+ years unweighted			0·89	1·94	2·91	2·98	0·22	−4·88
CC1 all 1+ years weighted			2·00	1·08	3·25	3·25	0·34	−5·41
CB	1	29	1·89	0·72	3·05	3·10	0·29	−5·14
CB	2	12	2·02	1·11	2·48	2·79	0·23	−4·53
CB	3	1	2·10	1·53				
CB	4	0						
CB all 1+ years unweighted			1·93	0·85	2·96	3·05	0·32	−5·04
CB all 1+ years weighted			2·01	1·12	3·69	3·74	0·49	−6·39
CC2	1	32	1·88	0·89	2·90	3·03	0·24	−4·93
CC2	2	13	2·03	1·23	2·28	3·50	0·24	−5·96
CC2	3	0						
CC2	4	0						
CC2 all 1+ years unweighted			1·93	0·89	2·71	2·93	0·28	−4·77
CC2 all 1+ years weighted			1·96	0·97	2·60	2·60		−4·12

LI, Little Indian Creek; CC1, upper Clear Creek; CB, Conards Branch and Richland Creek; CC2, lower Clear Creek; n, number of individuals; M_B, mass (g); L_T, length (mm); S.D., from the predictive regression line of $\log_{10} M_B$ on $\log_{10} L_T$.

comparison (Fig. 4). Creek chub from Manitoba grew slower than in the present study but perhaps lived slightly longer (Moshenko & Gee, 1973). Creek chub are sexually dimorphic, so combining male and female creek chub data for routine fish population analysis can make it impossible to detect subtle adverse effects, especially if the contaminant in question impacts the sexes differently as PCBs appear to do.

In ideal conditions, male creek chub continue to grow in L_T and M_B as they age, although the rate of mass gain outpaces gains in L_T as fish age. Although female creek chub continued to gain mass as they aged, gains in L_T were not as pronounced as they were in males. This was probably due to the physiological energy demands of egg production. When the M_B and L_T relationships were examined with the sexes combined, no trends in the data were observed. The relationships for males and females analysed separately, however, indicated the possibility of differences between PCB contaminated creek chub and reference site creek chub. Generally, a functional regression value of b equal to 3·0 indicates isometric growth. Higher values of b indicate better 'condition' or a more rotund fish (Ricker, 1975; Anderson & Gutreuter, 1989). Lower values of b would indicate less mass being added by the fish as length increased. In extreme cases, low b values would indicate emaciation. In this study, functional b for reference site males within age classes 1+ to 3+ years, was *c.* 3·0 (range: 3·01–3·04), whereas for reference females within age classes 1+ to 3+ years, the functional b range was somewhat higher (3·10–3·26). Combining these clearly different mass and length relationships gives ample opportunity for masking subtle growth changes. At the most contaminated site (CC1), male fish in age classes 1+ up to 3+ years were heavier for the same L_T than reference males (b range: 3·20–3·43). Not only was this indicative of a growth enhancement, but it also appeared to obscure the sexual dimorphism in creek chub. By contrast, female fish in age classes 1+ up to 3+ years at CC1 had a decreased M_B and L_T relationship (b range: 1·48–2·88) compared to 1+ up to 3+ year females at the reference site, or any other site, or even compared to males of those age classes at any other site. In older age classes (3+ and 4+ years) functional b appeared to decline more quickly at the most PCB contaminated sites, especially in females. This appears to be a subtle indicator of the 'wasting' syndrome reported by others (Halter & Johnson, 1974; Bengtsson, 1980; Cleland *et al.*, 1988). The year to year growth comparison is made more difficult by the loss of the older age classes at the contaminated sites [loss of age 5 years in males, and ages 4 and 5 years (and even age 3 years at CC2) in females].

Although the data show a trend toward larger size of the 1 year old fish from the PCB-exposed populations, this trend disappears beyond 2 years of age. Beyond age 2 years, both L_T and M_B of male creek chub were reduced at the contaminated sites when compared to males from the reference site. Several studies have shown that growth of larvae and juvenile fishes can be enhanced at low doses of PCBs (Bengtsson, 1979, 1980). The lower growth rate in younger age class males at the LI reference site compared to the males at the contaminated sites (CC1 in particular), however, could possibly be related to increased competition due to species diversity and the presence of piscivorous fishes. Katz & Howard (1955) documented that creek chub grew better in a polluted but recovering zone of Lytle Creek, OH, U.S.A. rather than upstream of the sewage discharge. Although the types of contaminants present were not specified, nutrient enrichment and a lack of competition and predators probably contributed to this increased growth. No fish larvae were ever observed at CC1, and no creek chub spawning activities were ever observed at either Clear Creek site (unpubl. obs.). It is notable that the loss of the older age classes was more pronounced for females compared to males. Thus, while PCB contamination seemed to affect both ends of the age spectrum (reduced or no spawning, enhanced growth at early ages, wasting and loss in the older age classes), the severity of the impact for each part of the age spectrum was gender-specific.

The creek chub makes an excellent field study species since it is relatively abundant and present in many PCB contaminated streams. Laboratory studies have shown growth to be a sensitive endpoint in PCB feeding studies of fish, but these effects are not as easily observed in field studies. Despite the large sample size, many other variables may obscure the subtle effects of PCBs on growth. Even though no spawning was observed at CC1, creek chub were probably present due to in-migration from nearby uncontaminated tributaries. Creek chub have been observed to rapidly repopulate areas (Olmsted & Cloutman, 1974). The immigration of young fish does complicate the interpretation of these field observations. Nevertheless, it is expected that most of the fish captured within these study areas had spent the majority of their lives at these locations. Another complicating factor is that rarely has an attempt been made to compare current fish densities of streams to what is expected. Larimore (1955) documented the presence of >28 kg ha^{-1} of four minnow species [*Campostoma anomalum* (Rafinesque), *Ericymba buccata* cope, *Pimephales notatus* (Rafinesque) and *Semotilus atromaculum* (Mitchill)] from a small stream in Illinois, U.S.A. This equates to *c.* 17 000 fish from a 500 m stretch of stream. There was no attempt to evaluate fish densities. Unless there is widespread indiscriminate mortality that either significantly alters fish species diversity in a stream or destroys the population, subtle adverse effects may be hard to document.

Despite the difficulty in detecting contaminant effects using classic fish ecology methods, trends are apparent in the PCB growth enhancing effects in fish prior to 24 months of age and the ultimate reduction of growth and survival in creek chub when sexes are analysed separately. These PCB study areas have reduced age 0 and > age 3 years fish which is consistent with Bengtsson's (1979) observation that juveniles are more durable than older and younger fishes (larvae and fry). The early mortality of older females from the PCB-contaminated sites is also consistent with observations made by Black *et al.* (1998*b*) in mummichog from New Bedford Harbor. Even though the present results on age structure and growth variables at these PCB-contaminated sites are consistent with other laboratory and field research, these routine field evaluation methodologies do not indicate what may be happening in the environment. To determine the effects of PCBs on native fish populations more fully, an in-depth investigation into physiological health of individuals of the population is imperative.

We would like to thank J. Exl and M. Litwin for their assistance with field collections. This work was funded by the U.S. Fish and Wildlife Service's Environmental

Contaminants programme and the Department of the Interior's Natural Resource Damage Assessment and Restoration programme.

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ATTACHMENT 4

Neal's Landfill Fish Sampling Data
November 2005

TABLE 1
Neal's Landfill
Fish Sample Data
November 2005
FINAL

CBS SAMPLE ID	LAB SAMPLE ID	LOCATION	SAMPLE DATE	FISH TYPE	Total PCB Congeners	Validation Flags	dilution factor	% LIPIDS	LENGTH	WEIGHT	NUMBER	TYPE	COMMENTS	Gender	Age	QA/QC
NL-0001	G532-6-1	NL-2, Richland Creek	11/09/05	Longear Sunfish	**878,433**		50	2.20	125	39	1	Whole Body		M		
NL-0002	-2	NL-2, Richland Creek	"	Longear Sunfish	**631,035**		50	1.47	112	30	1	Whole Body		M		
NL-0003	-3	NL-2, Richland Creek	"	Longear Sunfish	**249,901**		20	1.70	137	47	1	Whole Body		M	5+	
NL-0004	-4	NL-2, Richland Creek	"	Longear Sunfish	**1,429,500**		100	2.79	125	41	1	Whole Body		M		
NL-0005	-5	NL-2, Richland Creek	"	Creek Chubs	**238,082**		20	1.00	156	36	4	Whole Body	Composite	4M		TT Split
NL-0005 DUP	-6				**242,687**		20	0.84								
NL-0006	-7	NL-2, Richland Creek	"	Creek Chubs	**76,708**		10	0.78	161	37	1	Whole Body		F		
NL-0007	-8	NL-2, Richland Creek	"	Creek Chubs	**115,573**		10	0.90	168	50	1	Whole Body		F	2+	
NL-0008	-9	NL-2, Richland Creek	"	Creek Chubs	**89,378**		10	0.80	149	30	1	Whole Body		M		
NL-0009	-10	NL-2, Richland Creek	"	Creek Chubs	**244,821**		20	1.34	153	39	1	Whole Body		M		
NL-0010	-11	NL-2, Richland Creek	"	Creek Chubs	**670,404**		50	1.58	147	30	1	Whole Body		M		
NL-0011	-12	NL-2, Richland Creek	"	Creek Chubs	**339,025**		10	0.94	147	30	1	Whole Body		M		
NL-0013	-13	NL-2, Richland Creek	11/10/05	Creek Chubs	**218,136**		20	1.10	177	58	1	Whole Body		F	2+	
NL-0013 DUP	-14				**209,509**		20	0.93								
NL-0014	-15	NL-2, Richland Creek	"	Creek Chubs	**324,378**		20	0.76	151	33	1	Whole Body		M		
NL-0015	-16	NL-2, Richland Creek	"	Creek Chubs	**214,507**		20	0.58	148	30	1	Whole Body		F		
NL-0016	-17	NL-2, Richland Creek	"	Longear Sunfish	**621,348**		50	1.05	111	30	1	Whole Body		M		
NL-0017	-18	NL-2, Richland Creek	"	Longear Sunfish	**591,967**		50	1.03	121	33	1	Whole Body		M		
NL-0018	-19	NL-2, Richland Creek	"	Longear Sunfish	**210,490**		20	1.03	118	31	1	Whole Body		M		
NL-0019	-20	NL-2, Richland Creek	"	Longear Sunfish	**237,290**		20	0.38	132	44	1	Whole Body		M	5+	
NL-0020	-21	NL-2, Richland Creek	"	Longear Sunfish	**92,288**		10	0.34	114	30	1	Whole Body		F		
NL-0021	-22	NL-2, Richland Creek	"	Longear Sunfish	**657,998**		50	1.30	106	23	4	Whole Body	Composite	3M / 1F		TT Split
NL-0021 DUP	-23				**522,359**		50	1.19								
NL-0022	-24	NL-1, Conard's Branch	11/10/05	Creek Chubs	**1,981,867**		100	0.54	150	30	1	Whole Body		F		
NL-0023	-25	NL-1, Conard's Branch	"	Creek Chubs	**1,924,949**		100	0.38	157	34	1	Whole Body		M	2+	
NL-0024	-26	NL-1, Conard's Branch	"	Creek Chubs	**1,611,173**		100	0.44	165	36	1	Whole Body		M	2+	
NL-0025	-27	NL-1, Conard's Branch	"	Creek Chubs	**1,132,426**		50	0.40	139	27	1	Whole Body		F		
NL-0026	-28	NL-1, Conard's Branch	"	Creek Chubs	**1,035,380**		50	0.73	127	21	1	Whole Body		M		
NL-0027	-29	NL-1, Conard's Branch	"	Creek Chubs	**1,615,779**		100	0.55	130	22	1	Whole Body		M		
NL-0028	-30	NL-1, Conard's Branch	"	Creek Chubs	**5,287,822**		200	0.67	132	21	1	Whole Body		M		
NL-0029	-31	NL-1, Conard's Branch	"	Creek Chubs	**2,923,426**	J	100	0.51	129	20	1	Whole Body		M		
NL-0030	-32	NL-1, Conard's Branch	"	Creek Chubs	**1,730,340**		100	0.51	132	20	1	Whole Body		F		
NL-0031	-33	NL-1, Conard's Branch	"	Creek Chubs	**3,505,322**	J	100	0.47	125	18	4	Whole Body	Composite	4M		TT Split

Notes:
PCBs in pg/g wet weight
Lipids in % wet weight
length in mm
weight in grams
For composite samples, the length and weight listed is the mean
nm = not measured
J=estimate due to QC issues
Dilutions of 50x, 100x and 200x were externally diluted
Total Congener results for externally diluted analyses are NOT adjusted for extraction standard recovery

TABLE 1a
Neal's Landfill Fish Sampling
November 2005
WHO Congener Data
FINAL

Sample Date	Fish Type	Stream Reach	Sample Number	WHO Congener (pg/g) 77	81	105	114	118	123	126	156/157	167	169	189	Mammalian
			Mammalian TEF:	0.0001	0.0001	0.0001	0.0005	0.0001	0.0001	0.1	0.0005	0.00001	0.01	0.0001	WHO-TEQ
11/09/05	Longear Sunfish	NL-2, Richland Creek	NL-0001	5070	265	37100	3050	74600	2580	116	2480	980	24.7	99.5	26.6
11/09/05	Longear Sunfish	NL-2, Richland Creek	NL-0002	3990	252	29700	2380	52400	2030	84.4	1950	675	4.8	66.6	19.5
11/09/05	Longear Sunfish	NL-2, Richland Creek	NL-0003	749	51.2	12600	1110	25100	996	38.8	1150	440	9.65	62.3	9.1
11/09/05	Longear Sunfish	NL-2, Richland Creek	NL-0004	9570	438	45200	3430	78500	2380	127	2220	769	49.65	77.2	29.6
11/09/05	Creek Chubs	NL-2, Richland Creek	NL-0005	1490	95.7	10200	763	18100	687	34.6	669	273	9.85	34.8	7.3
11/09/05	Creek Chubs	NL-2, Richland Creek	NL-0005 DUP	1510	91.6	9420	714	18300	558	42.5	671	275	9.8	36.6	8.0
11/09/05	Creek Chubs	NL-2, Richland Creek	NL-0006	442	26.6	3170	250	5650	210	13.8	250	110	14.1	15.7	2.7
11/09/05	Creek Chubs	NL-2, Richland Creek	NL-0007	643	37.1	6470	520	11300	352	28.2	595	235	24	33.5	5.5
11/09/05	Creek Chubs	NL-2, Richland Creek	NL-0008	481	26.6	4090	325	6790	261	20.2	329	140	0.4905	16	3.5
11/09/05	Creek Chubs	NL-2, Richland Creek	NL-0009	1040	75	7720	643	13600	532	29.5	609	237	9.45	26.9	6.0
11/09/05	Creek Chubs	NL-2, Richland Creek	NL-0010	5550	272	17800	1300	31300	1290	76.7	968	363	24.6	41.3	14.7
11/09/05	Creek Chubs	NL-2, Richland Creek	NL-0011	2050	119	14500	1200	25600	1160	57.1	1050	383	24.75	41.8	11.4
11/10/05	Creek Chubs	NL-2, Richland Creek	NL-0013	1500	80	7940	787	16700	663	41.9	725	309	19.6	43.1	7.8
11/10/05	Creek Chubs	NL-2, Richland Creek	NL-0013 DUP	1400	83.4	7050	725	15900	633	42.4	681	294	9.6	35.3	7.6
11/10/05	Creek Chubs	NL-2, Richland Creek	NL-0014	2080	126	13800	999	26100	883	62.2	953	363	9.55	44.7	11.6
11/10/05	Creek Chubs	NL-2, Richland Creek	NL-0015	1370	108	8280	616	14400	577	35.8	549	237	9.8	28.7	6.7
11/10/05	Longear Sunfish	NL-2, Richland Creek	NL-0016	4450	247	30100	2280	46300	1840	103	1820	643	51.8	70.5	21.2
11/10/05	Longear Sunfish	NL-2, Richland Creek	NL-0017	3210	110	31900	2770	46000	2800	54.9	2240	797	58.6	91.9	17.0
11/10/05	Longear Sunfish	NL-2, Richland Creek	NL-0018	914	52.4	10200	918	20600	857	26.4	855	381	30.1	44.8	7.1
11/10/05	Longear Sunfish	NL-2, Richland Creek	NL-0019	891	42.4	17700	1470	31000	1290	27.3	1580	530	32.8	65	9.7
11/10/05	Longear Sunfish	NL-2, Richland Creek	NL-0020	826	35.9	3290	255	5460	218	10.0	227	93.4	0.4905	10.7	2.2
11/10/05	Longear Sunfish	NL-2, Richland Creek	NL-0021	5660	344	29700	2590	49400	2110	119	2010	676	5.1	69.7	23.0
11/10/05	Longear Sunfish	NL-2, Richland Creek	NL-0021 DUP	4440	288	24100	2160	42300	1640	98.1	1660	561	0.555	64.1	19.0
11/10/05	Creek Chubs	NL-1, Conard's Branch	NL-0022	12000	889	78400	5140	153000	3540	248	3670	1690	84.3	126	54.9
11/10/05	Creek Chubs	NL-1, Conard's Branch	NL-0023	17100	1330	87700	7590	162000	4460	339	4890	1820	49.2	171	67.9
11/10/05	Creek Chubs	NL-1, Conard's Branch	NL-0024	11300	409	71100	5710	129000	3750	261	3520	1330	4.57	116	52.3
11/10/05	Creek Chubs	NL-1, Conard's Branch	NL-0025	8000	508	42900	4550	77800	3330	126	2460	863	0.48	70.3	29.4
11/10/05	Creek Chubs	NL-1, Conard's Branch	NL-0026	6300	377	16600	1240	29000	1180	90.3	819	330	5.55	35.8	15.5
11/10/05	Creek Chubs	NL-1, Conard's Branch	NL-0027	11100	842	67700	4880	130000	4050	251	3540	1340	5.4	124	50.8
11/10/05	Creek Chubs	NL-1, Conard's Branch	NL-0028	29900	2360	111000	8520	201000	6550	439	4050	1710	0.535	159	85.3
11/10/05	Creek Chubs	NL-1, Conard's Branch	NL-0029	20100	1340	84200	5580	144000	4230	341	3660	1240	60.5	106	64.7
11/10/05	Creek Chubs	NL-1, Conard's Branch	NL-0030	9920	898	70100	5310	135000	4020	223	4050	1510	60	121	49.6
11/10/05	Creek Chubs	NL-1, Conard's Branch	NL-0031	19800	1180	75300	5860	149000	3360	313	3630	1410	4.56	119	61.0